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Moab Site

Site Observational Work Plan for the Moab, Utah, Site

Volume I

December 2003

Prepared for U.S. Department of Energy, Grand Junction, Colorado
under DOE Contract Number DE–AC13–02GJ79491.
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Acronyms and Abbreviations

ACL	alternate concentration limit
AEC	Atomic Energy Commission
ASTM	American Society for Testing and Materials
bgs	below ground surface
BLRA	baseline risk assessment
CDSA	characterization and disposal site alternatives
CERC	Columbia Environmental Research Center
CFR	*Code of Federal Regulations*
cfs	cubic feet per second
cm/s	centimeters per second
COC	contaminants of concern
COPC	contaminants of potential concern
CV	coefficient of variation
DAF	dilution attenuation factor
DF	Dupuit Forchheimer
DOE	U.S. Department of Energy
EIS	environmental impact statement
EPA	U.S. Environmental Protection Agency
ERA	ecological risk assessment
ESL	Environmental Sciences Laboratory
ET	evapotranspiration
EVS	Environmental Visualization
ft	foot (feet)
ft/day	foot (feet) per day
ft^2	square feet
ft^2/day	square feet per day
FEIS	Final Environmental Impact Statement
FEE	feasible, economical
FEU	feasible, uneconomical
FONSI	finding of no significant impact
gal	gallon(s)
GCAP	Ground-Water Compliance Action Plan
GJO	Grand Junction Office
G/cm^3	grams per square centimeter
gpm	gallons per minute
HEW	U.S. Department of Health, Education, and Welfare
HI	hazard index
HQ	hazard quotient
i.d.	inside diameter
in.	inch
IR	infrared
K_d	distribution coefficient
kg	kilogram
lb/gal	pounds per gallon
L	liter(s)
L/s	liter(s) per second
LAI	leaf area index

m/day	meters per day
MAP	management action process
MCL	maximum concentration limit
µS/cm	microsiemens per centimeter
µm	micrometer
mg/Kg	milligrams per kilogram
mg/L	milligrams per liter
mL	milliliters
mL/g	milliliters per gram
µg/L	micrograms per liter
µm	micrometer
mm	millimeters
Moab Site	Moab Uranium Mill Tailings Site
mS/m	millisiemens per meter
NAO	North American Datum
NAS	National Academy of Sciences
NAV	North American Vertical Datum
NEPA	National Environmental Policy Act
NPS	National Park Service
NOI	Notice of Intent
NRC	U.S. Nuclear Regulatory Commission
NTN	no treatment necessary
o.d.	outside diameter
ORNL	Oak Ridge National Laboratory
OCS	Opposed Crystal System
ORP	oxidation-reduction potential
pCi/L	picocuries per liter
pCi/g	picocuries per gram
PEIS	Programmatic Environmental Impact Statement
PVC	polyvinyl chloride
RAP	remedial action plan
RBC	risk based concentration
rpm	revolutions per minute
R_d	distribution ratio
RRM	residual radioactive material
SDWA	Safe Drinking Water Act
SOWP	site observational work plan
SRK	Steffen, Robertson and Kirsten
TAGR	*Technical Approach to Ground Water Restoration*
TDS	total dissolved solids
U.S.C.	*United States Code*
USF&WS	U.S. Fish and Wildlife Service
UMTRA	Uranium Mill Tailings Remedial Action (Project)
UMTRCA	Uranium Mill Tailings Radiation Control Act
UNF	unfeasible
USGS	U. S. Geological Survey
VWP	vibrating wire piezometers

1.0 Introduction

1.1 Purpose and Scope

The purpose of this Site Observational Work Plan (SOWP) is to present the technical information necessary to allow selection of a ground water compliance strategy for the Moab Uranium Mill Tailings Site (Moab site). The Moab site is a former uranium-ore processing facility located about 3 miles northwest of the city of Moab in Grand County, Utah (Figure 1–1 and Figure 1–2), and lies on the west bank of the Colorado River at the confluence with Moab Wash. As required by the Floyd D. Spence National Defense Authorization Act for Fiscal Year 2001 (the act), the title of the Moab site was transferred to the U.S. Department of Energy (DOE) along with the responsibility for site cleanup in accordance with Title I of the Uranium Mill Tailings Radiation Control Act (UMTRCA). The act further requires that remediation of the site include ground water restoration. Consistent with requirements for ground water remediation at other Title I Uranium Mill Tailings Remedial Action (UMTRA) Ground Water Project sites, this SOWP is being prepared to assist in the selection of a ground water compliance strategy for the Moab site. This document will also serve as a ground water technical support document for the Environmental Impact Statement (EIS) for site remediation that is being prepared concurrently.

Ground water compliance standards applicable to the Moab site are the U.S. Environmental Protection Agency (EPA) standards established in Title 40 *Code of Federal Regulations* Part 192 (40 CFR 192). Subparts A and B of 40 CFR 192 provide standards for cleanup and final disposal of contaminated materials for Title I UMTRA Ground Water Project sites. Subpart A standards apply to protection of ground water from future contamination released from the disposal system after cleanup is complete. Subpart B standards pertain to cleanup of residual radioactive materials, including ground water, at the site. The Subpart B cleanup standards are the same as the Subpart A protection standards except that Subpart B allows for an extended time frame to attain compliance with the standards and allows the use of a natural flushing compliance strategy, providing certain other criteria are met. Section 2.2.1, “EPA Ground Water Standards,” provides a more detailed discussion of the standards in 40 CFR 192.

Surface remediation of the Moab site has not yet been initiated, as a final decision on the permanent disposition of contaminated surface materials has not been made. It is expected that the preferred surface alternative will be identified in the final EIS for the site. Unlike other Title I UMTRA Ground Water Project sites, it is likely that selection of a ground water compliance strategy will be made concurrently with, rather than following, selection of a proposed action for surface remediation. The decision on the surface cleanup alternative and which ground water standards are applicable could be important factors in determining the appropriate ground water compliance strategy. On the other hand, evaluation of ground water remedial alternatives could also have some bearing on the decision regarding surface remediation. Therefore, DOE will evaluate ground water compliance alternatives for the potential range of final surface conditions (i.e., no action, on-site tailings disposal, or off-site tailings disposal; see Section 6.0 for further discussion).

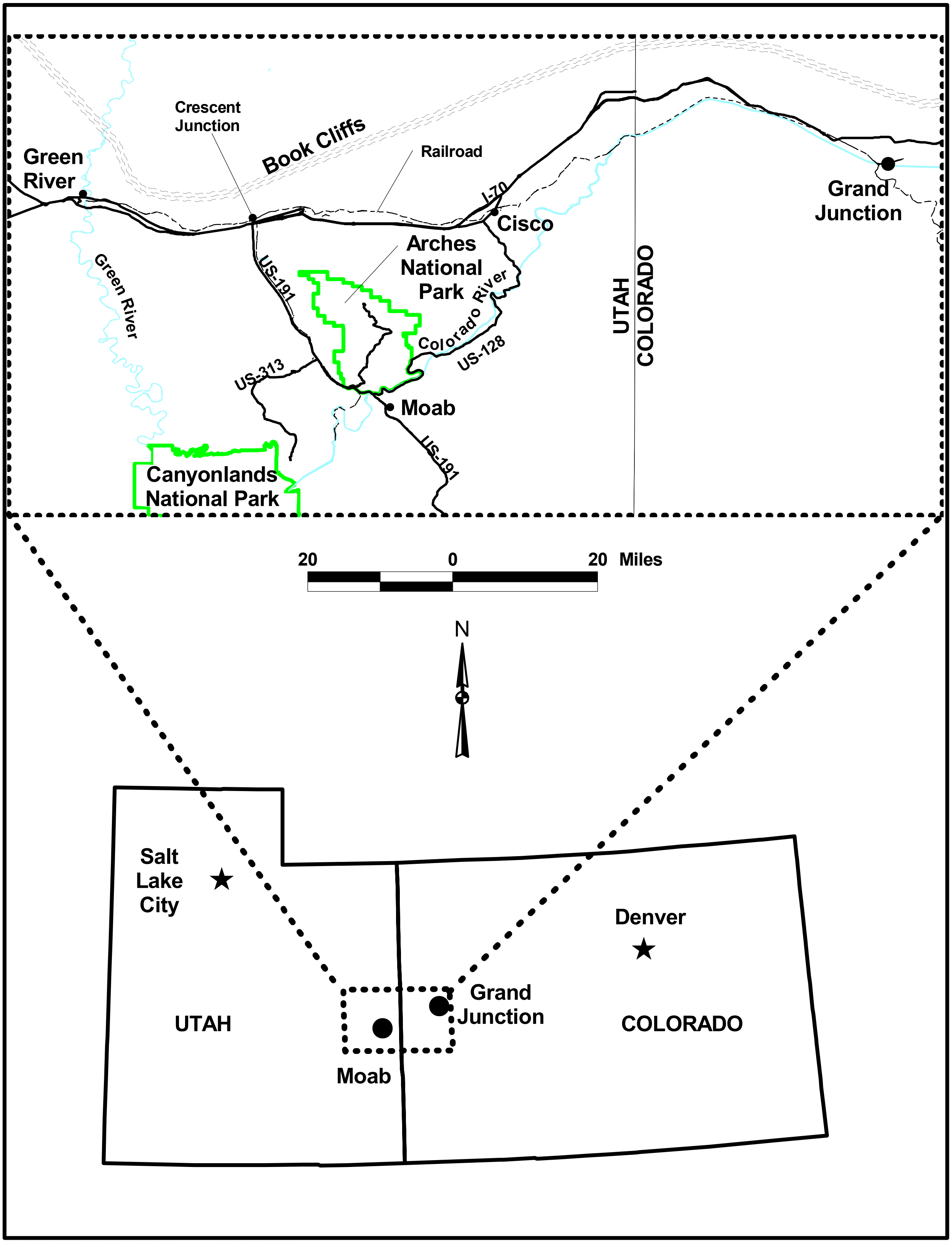

Figure 1–1. Regional Location Map

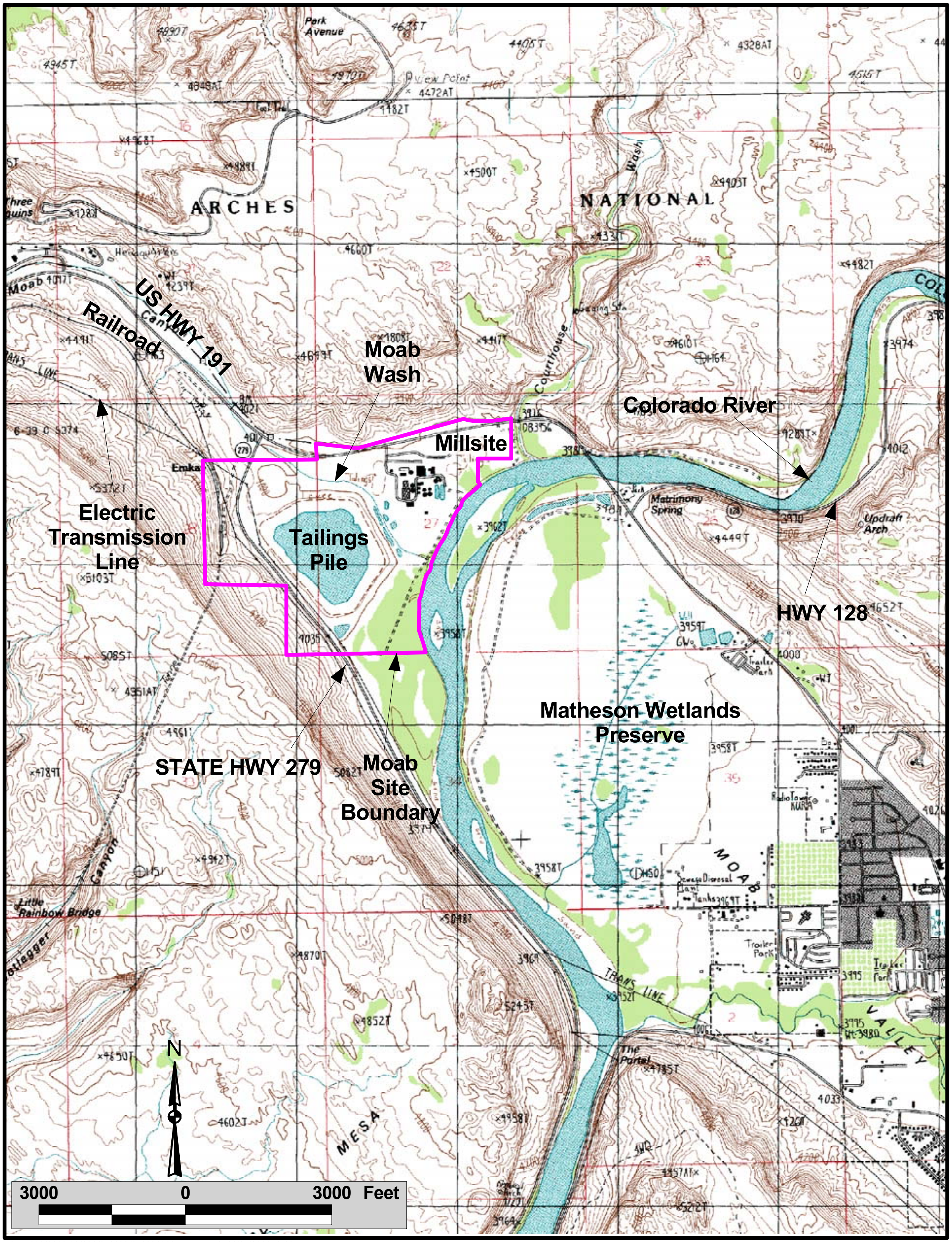

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Figure 1–2. Moab Site and Surrounding Area (modified from 1983 USGS topographic map)

In October 1996, DOE issued the *Final Programmatic Environmental Impact Statement for the Uranium Mill Tailings Remedial Action Ground Water Project* (PEIS) (DOE 1996). The proposed action (DOE's preferred alternative) in the PEIS is to use a consistent, risk-based decision process that results in a ground water compliance strategy tailored for each UMTRA Ground Water Project site. The framework for the decision process is discussed in Section 2.3.

DOE's goal is to implement a cost-effective compliance strategy for ground water that is protective of human health and the environment. The range of potential ground water alternatives includes (1) no ground water remediation, (2) natural flushing of ground water, and (3) active remediation of ground water (see further discussion in Section 2.0). Different alternatives or combinations of alternatives may be applicable to different contaminants in ground water or to different portions of the affected aquifer. Descriptions and evaluations of characterization data that support the recommended compliance strategy are presented in this document.

Compliance requirements for meeting the regulatory standards for ground water at the Moab site are presented in Section 2.0. Site background information, including an overview and history of the former milling operation and previous investigations are reviewed in Section 3.0. Results of DOE field investigations conducted at the site are presented in Section 4.0. Recent and historical site-specific characterization of the geology, hydrology, geochemistry, and ecology are evaluated in Section 5.0 and synthesized in the site conceptual model in Section 6.0. Results of the flow and transport modeling for ground water are summarized in Section 7.0, and the proposed compliance strategy for ground water cleanup is presented in Section 8.0. A screening level analysis of ground water remediation alternatives is presented in Section 9.0, and limitations to the conceptual site model are described in Section 10.0.

1.2 UMTRA Ground Water Project Programmatic Documents

Programmatic documents that guide the SOWP include the *UMTRA Ground Water Management Action Process* (DOE 2002f), the *Technical Approach to Ground Water Restoration* (DOE 1993b), the *Technical Approach Document* (DOE 1989), and the PEIS (DOE 1996). The *Management Action Process* document states the mission and objectives of the UMTRA Ground Water Project and provides a technical and management approach for conducting it. Technical guidelines for conducting the ground water program are presented in the *Technical Approach to Ground Water Restoration* document; the *Technical Approach Document* considers disposal cell design and protection of ground water. DOE will follow PEIS guidelines to assess the potential programmatic impacts of the UMTRA Ground Water Project, to determine site-specific ground water compliance strategies, and to prepare site-specific environmental impact analyses more efficiently. The PEIS provides a consistent, risk-based, decision-making framework for selecting an appropriate compliance strategy. This framework is discussed in Section 2.3 and illustrated in Figure 2−1.

1.3 Site-Specific Documents Applicable to the SOWP

Numerous historical documents have been prepared for the Moab site; most of these were completed to fulfill different regulatory requirements associated with site licensing, site operation, site monitoring, and more recently, site reclamation. Some reports addressed specific concerns about particular issues (e.g., several reports were prepared in response to U.S. Nuclear Regulatory Commission [NRC] review comments on site reclamation plans). The most notable studies are described in Section 3.0 of this document regarding previous investigations.

In addition to historical documents, several site-specific documents are currently being prepared, and others are being planned to support remedial action decisions for which DOE is now responsible under Title 1 of UMTRCA. As noted, analyses to support both surface remediation and ground water remediation decisions are being performed concurrently. A human health risk assessment and a biological assessment have been prepared as appendixes to support the EIS. Portions of these documents relative to ground water compliance strategy selection are summarized in the SOWP. Other smaller studies or analyses are also being documented in calculation sets, many of which are included as appendixes in Volume III of this SOWP. Some of these are relevant to ground water remediation; others pertain more to surface remediation. The evaluation of ground water compliance strategies in Section 8.0 of this SOWP will draw upon all pertinent historical documents and those being prepared concurrently with the SOWP. All these sources of data and information are used to support completion of the EIS for remediation of the Moab site. The Record of Decision for the EIS will document the surface disposal and ground water alternative decision. The compliance strategy for surface cleanup and ground water will be provided together in a separate remedial action plan that will provide more specific information about remediation technologies and system design.

End of current text

2.0 Regulatory Framework

This section identifies the regulatory framework to be applied to the selected ground water compliance strategy at the Moab site to achieve compliance with the EPA standards in 40 CFR 192 and the final rule to the standards published in the *Federal Register* at 60 FR 2854.

2.1 Requirements of the Floyd D. Spence Act

Remediation of the Moab site is mandated by the Floyd D. Spence National Defense Authorization Act for Fiscal Year 2001 (House of Representatives 2000). The act specifies that the license issued by NRC for the materials at the Moab site be terminated and that the title and responsibility for cleanup be transferred to DOE. The act further designates that the Moab site undergo remediation in accordance with Title 1 of UMTRCA (42 *United States Code* [U.S.C.] 7901), with certain exemptions.

The act also specified that DOE "…prepare a plan for remediation, including ground water restoration, of the Moab site in accordance with Title I of the Uranium Mill Tailings Radiation Control Act of 1978…" The act further required the Secretary of Energy to obtain from the National Academy of Sciences (NAS) "...the technical advice, assistance, and recommendations of the National Academy of Sciences in objectively evaluating the costs, benefits, and risks associated with various remediation alternatives, including removal or treatment of radioactive or other hazardous materials at the site, ground water restoration, and long-term management of residual contaminants" (House of Representatives 2000). DOE subsequently completed the draft *Preliminary Plan for Remediation* (DOE 2001a). After reviewing the draft plan, NAS provided a list of recommendations on June 11, 2002, for DOE to consider during its assessment of remediation alternatives for the Moab site. DOE does not intend to finalize a separate plan for remediation, but instead will incorporate information from the plan with the EIS and will use the EIS process to support decision-making for remediation of the Moab site.

2.2 Uranium Mill Tailings Radiation Control Act

The U.S. Congress passed UMTRCA in 1978 in response to public concerns about potential health hazards from long-term exposure to uranium mill tailings. UMTRCA authorized DOE to stabilize, dispose of, and control uranium mill tailings and other contaminated materials at inactive uranium-ore processing sites in a safe and environmentally sound manner.

Three UMTRCA titles apply to uranium-ore processing sites. Title I of UMTRCA designates inactive processing sites for remediation and stipulates that remedial action be selected and performed with the concurrence of NRC and in consultation with affected states and Indian tribes, directs NRC to license the disposal sites for long-term care, and directs DOE to enter into cooperative agreements with the affected states and Indian tribes. Title II applies to active uranium mills and directs NRC to regulate uranium mill tailings at those processing sites. Title III applies to certain uranium mills in New Mexico.

Title I directs DOE to complete remedial action at inactive uranium mill tailings sites at which all or a substantial portion of uranium was processed for sale to a federal agency, and which no longer had a license to process uranium ore as of January 1, 1978. The Secretary of Energy was given the authority to add sites to the list (DOE 1996). In 1988, Congress passed the Uranium Mill Tailings Remedial Action Amendments Act (42 U.S.C. Section 7922 et seq.), authorizing DOE to extend without limitation the time needed to complete ground water remediation at the Title I processing sites. Congress amended UMTRCA in 2000 to designate the Moab milling site as a processing site in accordance with Title I of UMTRCA.

2.2.1 EPA Ground Water Standards

As directed by UMTRCA, EPA published 40 CFR 192, "Health and Environmental Protection Standards for Uranium and Thorium Mill Tailings." The standards in 40 CFR 192, Subparts A, B, and C, apply to the remediation and final disposition of contaminated materials, including ground water, for Title I sites. Remediation of the Moab site must be in compliance with these standards.

The Subpart A standards for control of residual radioactive materials apply to disposal of these materials at processing or disposal sites and were established as a means of monitoring long-term performance of the disposal system. Subpart A provides numerical standards to be met for ground water protection from future contamination released from the disposal system after disposal is complete. Provisions are also made for the application of alternate concentration limits (ACLs) as the protection standards, where appropriate. Numerical ground water standards specified in 40 CFR 192 are provided in Table 2–1. Corrective actions are required within 18-months if contaminant concentrations in ground water at disposal sites exceed the ground water protection standards. Subpart A standards will apply to ground water associated with the long-term disposal of tailings from the Moab site, whether that disposal is on site or at a relocation site.

Table 2–1. EPA Ground Water Standards in 40 CFR 192

Constituent	MCL[a]
Arsenic	0.05 mg/L
Barium	1.0 mg/L
Cadmium	0.01 mg/L
Chromium	0.05 mg/L
Lead	0.05 mg/L
Mercury	0.002 mg/L
Molybdenum	0.10 mg/L
Nitrate (as N)	10 mg/L
Selenium	0.01 mg/L
Silver	0.05 mg/L
Radium-226+228	5 pCi/L[b]
Uranium-234+238	30 pCi/L[b]
Gross alpha	15 pCi/L[b]

[a] MCL = the maximum concentration limit in 40 CFR 192. The MCL for uranium of 30 picocuries per liter (pCi/L) is equal to 0.044 milligrams per liter (mg/L) if U-234 and U-238 are in equilibrium.

[b] pCi/L (picocuries per liter).

Subpart B standards for cleanup provide numerical standards for cleanup of residual radioactive materials based on concentrations of radium-226 in surface materials (e.g., soils) and for exposure to radiation in buildings. Numerical standards for the cleanup of ground water are the same as the protection standards specified in Subpart A. The cleanup standards also permit use of ACLs or supplemental standards as the appropriate cleanup standards based on site-specific circumstances. However, unlike Subpart A, an extended time frame is permitted for attaining Subpart B ground water cleanup standards. In addition to active remediation, natural flushing is an acceptable means of meeting the standards if they can be met within 100 years and if enforceable institutional controls can be put in place during that time.

Subpart C of 40 CFR 192 provides guidance for ensuring that provisions of Subparts A and B are met. Subpart C requires that conditions of Subparts A and B are met on a site-specific basis using information gathered during site characterization and monitoring. The approach to meet the conditions of Subparts A and B should be stated in the remedial action plan(s), including a consideration of ground water movement. If natural flushing is part of the ground water compliance strategy, Subpart C requires compliance monitoring and has requirements for points of compliance to verify anticipated plume movement and the associated reduction in plume contamination.

Subpart C specifies certain criteria under which DOE may apply supplemental standards to contaminated ground water in lieu of background levels, maximum concentration limits (MCLs) in 40 CFR 192, or ACLs identified in Subpart A. Supplemental standards may be applied if any of the following conditions are met:

- Remedial action necessary to implement Subpart A or B would pose a significant risk to workers or the public.
- Remedial action to meet the standards would directly produce environmental harm that is clearly excessive, compared to the health benefits of remediation, to persons living on or near the sites, now or in the future.
- The estimated cost of remedial action is unreasonably high relative to the long-term benefits, and the residual radioactive materials do not pose a clear present or future hazard.
- There is no known remedial action.
- The restoration of ground water quality at a designated processing site is technically impracticable from an engineering standpoint.
- The ground water meets the criteria of limited-use ground water. Subpart B of 40 CFR 192 defines limited-use ground water as ground water that is not a current or potential source of drinking water because at least one of the following conditions is present: (1) the concentration of total dissolved solids (TDS) exceeds 10,000 milligrams per liter (mg/L); (2) widespread, ambient contamination is present that cannot be cleaned up using treatment methods reasonably employed in public water systems; (3) the quantity of water available to a well is less than 150 gallons per day (0.1 gpm). When limited-use ground water applies, supplemental standards ensure that current and reasonably projected uses of the ground water are preserved.
- 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 residual radioactive materials.

2.3 National Environmental Policy Act

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

During the UMTRA Project, DOE prepared site-specific NEPA documentation (either an Environmental Assessment or an EIS) for each Title I uranium-ore processing site to address surface remediation at the site (i.e., cleanup of tailings, residual processing materials, soil, and buildings). An evaluation of the need for ground water remediation at these sites was deferred to a separate DOE project.

In 1994, DOE drafted the PEIS (DOE 1996), which was made final in 1996. The purpose of the PEIS was to present an analysis of the potential effects of implementing three programmatic alternatives for ground water compliance along with the "no action" alternative at the 22 designated processing sites evaluated in the UMTRA Ground Water Project. The preferred alternative for the UMTRA Ground Water Project was published in a Record of Decision in 1997. All subsequent actions within the UMTRA Ground Water Project comply with the Record of Decision.

For the Moab site, NRC prepared a draft EIS in 1996 to evaluate the environmental consequences of approving a license amendment application and allowing the Atlas Corporation to reclaim the existing tailings pile in place. Ground water cleanup was not considered as a separate component of the EIS; only the potential impacts of surface remediation on the ground water system were evaluated. This was a major concern raised during the public and agency review of the draft document (NRC 1999). The EIS was finalized in March 1999; however, a Record of Decision was never issued. Subsequent transfer of title of the site to DOE has also resulted in a transfer of NEPA compliance responsibilities from NRC to DOE.

DOE is currently preparing an EIS for remediation of the Moab site. A Notice of Intent was published (67 *Federal Register* 77969) and scoping was performed. Alternatives for surface remediation, ground water protection, and ground water remediation were evaluated through the EIS process. As stated in the Notice of Intent, evaluation of the ground water compliance strategies for Subpart B follows the framework presented in the PEIS (DOE 1996) and is consistent with the compliance strategy selection process followed for other Title I UMTRA Ground Water Project sites. However, because no preferred alternative for surface remediation had been selected at the time this SOWP was prepared, strategies for ground water cleanup (Subpart B) are identified for both the on-site and off-site disposal alternatives. For this reason, three different conceptual site models are developed in this SOWP: the no action baseline along with both on-site and off-site disposal models. Section 6.0 presents a detailed discussion of the conceptual site models.

The PEIS ground water compliance selection framework is presented in Figure 2–1. The framework takes into consideration human health and environmental risk, stakeholder input, and cost. A systematic approach is followed until one or a combination of two or three general compliance strategies is selected for each alternative evaluated. The following three compliance strategies are allowable under the regulations:

- ***No remediation***—Compliance with the EPA ground water cleanup standards would be met without altering the ground water or cleaning it up in any way. This strategy could be applied for those contaminants that are already at or below MCLs established in 40 CFR 192 or background levels. This strategy could also be used for naturally occurring (not milling-related) contaminants with concentrations above MCLs or background levels that qualify for supplemental standards or ACLs as defined in Section 2.2.1.
- ***Natural flushing***—Compliance with the EPA ground water cleanup standards would be met by allowing natural ground water movement and geochemical processes to decrease contaminant concentrations to regulatory limits. The natural flushing strategy could be applied where ground water compliance can be achieved within 100 years, where monitoring and effective institutional controls can be maintained, and where the ground water is not currently and is not projected to be a source for a public water system.
- ***Active ground water remediation***—Compliance with the EPA ground water cleanup standards cannot be met by the natural flushing or no further remediation strategies. This option requires application of engineered ground water remediation methods such as gradient manipulation, ground water extraction and treatment, land application, phytoremediation, or in situ ground water treatment to achieve compliance with the standards.

DOE is required by the PEIS to follow the ground water compliance selection framework summarized in Figure 2–1 in selecting the appropriate compliance strategy to clean up the uppermost aquifer affected by former milling activities at the Moab site. It is possible that different compliance strategies could apply to different constituents (e.g., natural flushing for one constituent and active remediation for another).

2.4 Other Regulations

Regulations other than those discussed above must be considered when evaluating ground water cleanup at the Moab site. Some of these regulations may be established at the federal level and others at the state level. The State of Utah's ground water quality regulations, besides establishing ground water standards, also provide requirements for ground water classification. The classification of ground water will, in part, determine what standards are applicable. At the federal level, EPA has similar guidelines for ground water classification (EPA 1988). The State of Utah ground water classification systems is provided in Table 2–2. Other state and federal regulations have standards that may or may not apply to ground water cleanup at the Moab site.

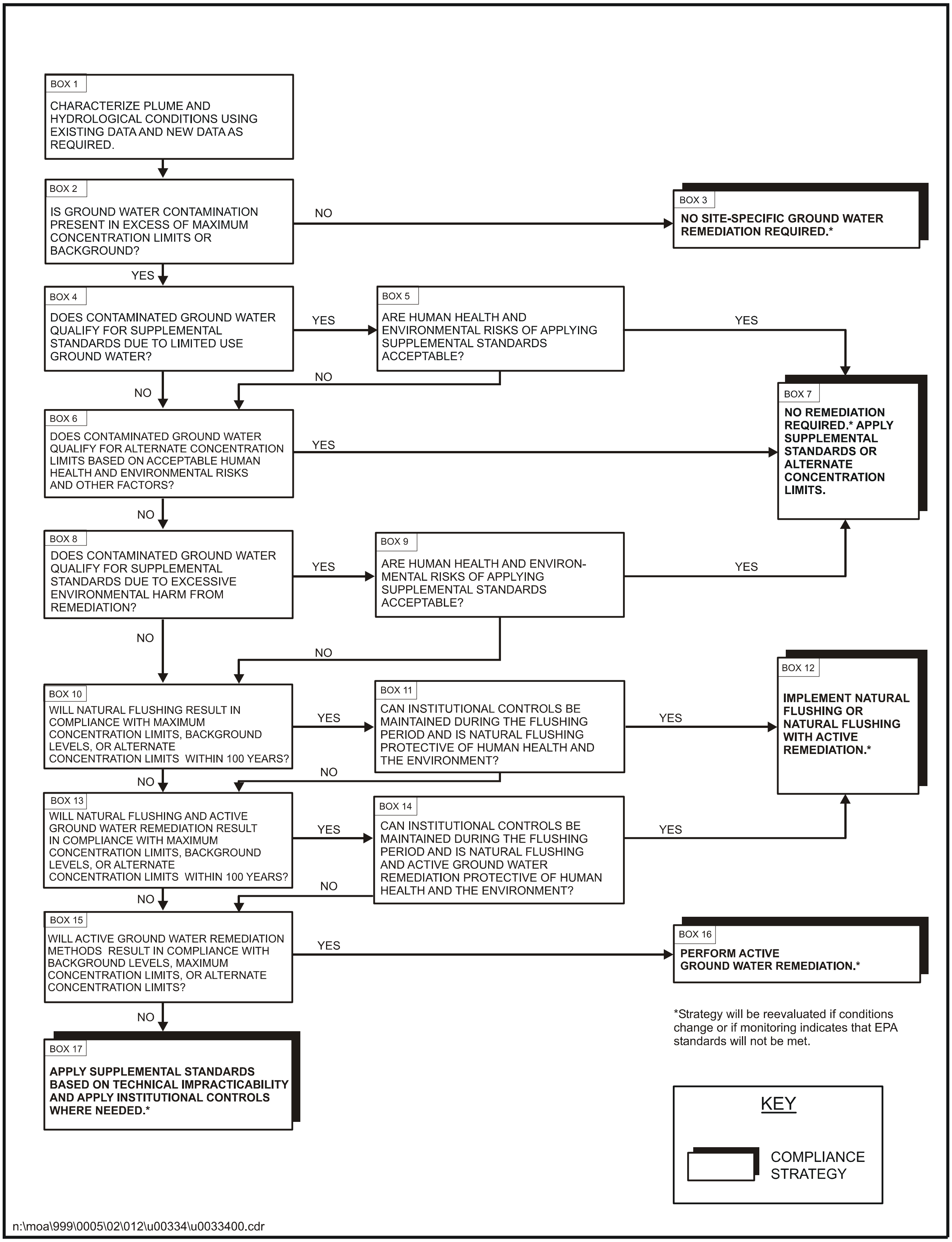

Figure 2–1. Summary of Ground Water Compliance Selection Framework

Table 2–2. State of Utah Ground Water Classification System

Class Number	Class Name	Class Requirements
IA	Pristine	TDS < 500 mg/L and no contaminants exceeding standards
IB	Irreplaceable	Source of water for a community public drinking water system for which no reliable supply of comparable quality is available
IC	Ecologically Important	Ground water is a source of ground water discharge important to the continued existence of wildlife habitat
II	Drinking-water quality	TDS >500 and <3,000 mg/L and no contaminants exceeding standards
III	Limited use	TDS >3,000 and <10,000 mg/L or one or more contaminants exceed standards
IV	Saline	TDS >10,000 mg/L

Like EPA standards, state of Utah regulations provide for the use of alternate ground water standards if ground water can have an effect on other water bodies. For example, establishing appropriate ground water cleanup standards may depend on the potential effects to hydraulically connected surface water. State of Utah ground water regulations provide for establishing concentrations in ground water “in order to meet applicable surface water standards” in certain instances (State of Utah Regulation R–317–6).

Other regulations may impose requirements besides establishing appropriate standards or selecting a remediation alternative. Though not directly providing guidelines for remediation alternative selection, the Endangered Species Act (16 U.S.C. 1531 et seq.) is an important regulatory driver for the remediation of the Moab site. Section 7 of the Endangered Species Act requires that every federal agency, in consultation with the Secretary of Interior, represented by the U.S. Fish and Wildlife Service (USF&WS), ensures that any action authorized by that agency is not likely to jeopardize the continued existence of any listed threatened or endangered species or its habitat. The USF&WS has determined that consultation is required because listed species and designated critical habitat are present at the site.

As discussed in the Notice of Intent to prepare an EIS for the Moab site, Executive Orders 11988 and 11990 mandate evaluation of federal actions taken in floodplains and wetlands and require federal agencies to issue regulations providing for public review of proposals or plans for such actions. DOE’s floodplain and wetlands regulations are codified at 10 CFR 1022, and DOE has completed a floodplain and wetlands assessment (which is included in the EIS) to comply with these regulations.

Any other state or federal regulations that are applicable to Moab site ground water remediation will be identified as the project progresses.

End of current text

3.0 Site History and Previous Investigations

This section describes the physical setting and history of the site and previous studies that have been undertaken in support of reclamation or remediation activities. Although much information is available for the Moab site, discussion here is confined to those aspects of the site that are pertinent to understanding the ground water system (including potential contaminant sources and interaction with surface water) and that may assist in selecting an appropriate ground water compliance strategy for the site. This section deals only with studies that were conducted before DOE assumed responsibility for the site; DOE-related investigations are presented in Section 4.0.

3.1 Site Setting

3.1.1 Physical Setting

The Moab site is located in the northern portion of a long, narrow, northwest-trending valley known as Spanish Valley; the northwest portion of the valley is also referred to as Moab Valley. The entire valley is within the Salt Anticline Section of the Colorado Plateau province, which includes the plateaus, mesas, and canyons of western Colorado, eastern Utah, northern Arizona, and northwestern New Mexico. The Salt Anticline Section is a topographic region characterized by great elongate depressions formed by removal of subterranean salt masses. Moab/Spanish Valley, Lisbon Valley, Salt Valley, and Castle Valley constitute the main salt-related valleys in the Salt Anticline Section. These northwest-trending valleys developed on collapsed or depressed anticlines with high surrounding walls. Doelling et al. (2002) provides detailed mapping and description of the Moab area geology, and Hunt (1956) provides a thorough discussion of the geology of the Colorado Plateau.

The floor of Moab Valley lies at an elevation of about 4,000 feet (ft) above mean sea level. Sandstone cliffs that form the valley walls near the former millsite site rise about 1,000 ft above the valley floor. The La Sal Mountains, about 12 miles east-southeast of the site, rise to elevations of more than 12,000 ft above mean sea level. A more detailed description of the physical characteristics of the site is provided in Section 5.0.

3.1.2 Site Description

The Moab site is irregularly shaped and encompasses approximately 400 acres; a 130-acre uranium mill tailings pile occupies much of the western portion of the site. The Moab site is bordered on the north and southwest by steep sandstone cliffs. The Colorado River forms the southeastern boundary of the site. U.S. Highway 191 (US-191) parallels the northern site boundary, and State Road 279 (SR-279) parallels the southwestern boundary. The entrance to Arches National Park is located less than 1 mile northwest of the site across US-191; Canyonlands National Park is about 12 miles to the southwest. The Union Pacific Railroad traverses a small section of the site just west of SR-279, then enters a tunnel and emerges several miles to the southwest. Moab Wash runs northwest to southeast through the center of the site and joins with the Colorado River. The wash is an ephemeral stream that flows only after precipitation or during snowmelt. Courthouse Wash, another ephemeral stream, but with a larger drainage than Moab Wash, discharges to the Colorado River about 300 ft east of the easternmost boundary of the site (Figure 1−2). Major site features are shown in Figure 1−2. The map in Figure 1−2 was completed in 1983; all but one of the on-site buildings have since been demolished, and water is no longer present on the surface of the tailings. However, surface contamination is still present in many areas of the site.

3.1.3 Meteorology and Climate

Hot summers and mild to cold winters characterize the high desert climate of Moab. January is the coldest month, with an average monthly temperature of about 30 °F, and July is the hottest month, with an average monthly temperature of about 81 °F. During the hot summer, approximately 25 days have temperatures of 100 °F or above (Pope and Brough 1996). Prevailing winds in the Moab area are from the west-southwest.

Average annual precipitation at Moab is 9 inches. The driest months are February and June, and the wettest months are April (spring) and October (early fall) when monthly averages of more than 1 inch of precipitation occur. Potential evapotranspiration (about 50 inches annually), potential or pan evaporation (about 60 inches annually), and lake evaporation (about 38 inches annually) all greatly exceed annual precipitation.

3.1.4 Land and Water Use

3.1.4.1 Land Use

Grand County encompasses approximately 2.4 million acres. According to the Utah Department of Community and Economic Development, the federal government owns 71.7 percent of the county land, the state owns 15.5 percent, 8.4 percent is American Indian tribal land, and 4.3 percent is private (GPU 2003).

Spanish Valley extends more than 13 miles southeast to northwest and averages about 1.5 miles in width. The amount of land suitable for cultivation is limited in the Moab area. The current land use pattern for private unincorporated land in Spanish Valley comprises 613 acres of irrigated agricultural and open space (excluding the Scott M. Matheson Preserve [Matheson Wetlands Preserve]), 2,030 acres of residential, commercial, and industrial property, and about 5,500 acres of vacant land (GPU 2003).

In the early 1990s, the Nature Conservancy purchased 875 acres of land across the river from Moab site, known as Moab Marsh (Figure 1−2), and designated it the Scott M. Matheson Wetlands Preserve. The Matheson Wetlands Preserve is jointly owned and managed by the Nature Conservancy and the Utah Division of Wildlife Resources. Portions of the Matheson Wetlands Preserve and private property adjacent to the preserve are grazed, primarily during the winter, by up to 50 head of cattle and 30 horses (NRC 1999).

The nearest residence is at the former Tex's Tour Center adjacent to the east side of the site. A river tours and gift shop business is located adjacent to the east side of Courthouse Wash at the easternmost boundary of the property. A restaurant and residence and two commercial parks for recreational vehicles, motor homes, and trailers are located along US-191 from 0.75 to 1.5 miles east of the site. The northwest edge of the main residential and commercial area of the city of Moab is about 1.8 miles from the site.

Arches National Park, directly adjacent to the site north of US-191, covers about 73,400 acres and is used extensively for tourism and recreation. The headquarters complex of Arches National Park is located in Moab Canyon about 1.2 miles northwest of the site. No other residences or residential areas are known to be located within 2 miles of the site (NRC 1999).

Adjacent land to the west, south, and north of the millsite is zoned Range and Grazing; the parcel on which the former Tex's Tour Center is located is zoned Resort/Commercial. The Moab site is zoned Specially Planned Area, a designation indicating that County approval must be obtained for any activity on the property.

3.1.4.2 Water Use

The Glen Canyon aquifer is the primary source of domestic water for the area and provides high quality drinking water for the city of Moab. Ground water is collected and piped to storage tanks, treated, and distributed. Private wells also obtain ground water from sandstones of the Glen Canyon aquifer. The nearest use of ground water is from a well completed in the Glen Canyon aquifer at Arches National Park headquarters.

The alluvial aquifer is a secondary source of domestic water for the Spanish Valley area. More than 200 alluvial wells are reported to be in Spanish Valley south of the site; most are used for irrigation, but some are also used for domestic water supply where recharge from the Glen Canyon aquifer dilutes the dissolved solids to acceptable levels (Eisinger and Lowe 1999).

Surface water withdrawn from the Colorado River is less than 39 cubic feet per second in Grand County. This water is used almost exclusively for agricultural irrigation. The river in the Moab area is used for swimming, rafting, boating, and fishing and is a recognized scenic waterway (NRC 1999). The nearest known Colorado River intake used for irrigation is at a ranch approximately 3 miles downstream of the Moab site (Dames & Moore 1974).

3.2 Site History and Operations

This section describes the operational history of the Moab site. Operations are broken into early and late operations; fundamental changes in site practices, particularly with regard to water use and effluent discharge, took place during this transition. A fundamental change in the focus of site characterization activities took place when the site went from operational status to standby and preparation for decommissioning and reclamation. A discussion of those later studies is presented in Section 3.5.

3.2.1 Early Operations (1956−1974)

The Moab site began uranium milling operations in October 1956. The mill area in 1953 before construction of the mill and in 1959 shortly after operations began is shown in Figure 3–1 and Figure 3–2, respectively. The mill was originally owned by the Uranium Reduction Company but was acquired by Atlas Corporation in 1962. According to Merritt (1971), the original uranium milling process used an acid leach circuit, which was changed to an alkaline process due to changes in ore composition. In 1967, an acid leaching and solvent extraction process was added to recover copper and vanadium as by-products. The acid-leach processing circuit was subsequently destroyed by fire in December 1968 (Atlas 1973).

All ore was brought to the site by truck. Because some ores contained commercial quantities of vanadium and copper in addition to uranium, they were segregated by type as received. According to Atlas (1973), ore was stored in 50- to 800-ton lots in the ore receiving area. The maximum storage in this area was estimated as 70,000 tons (Atlas 1973). Crushed ore was stored in stockpiles of up to 30,000 tons in each of various locations that are shown on Figure 3–3. Maximum storage of crushed ore was estimated at 100,000 tons (Atlas 1973).

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Figure 3–1. Pre-Mill Aerial Photo (1953)

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Figure 3–2. Atlas Millsite Photo (1959)

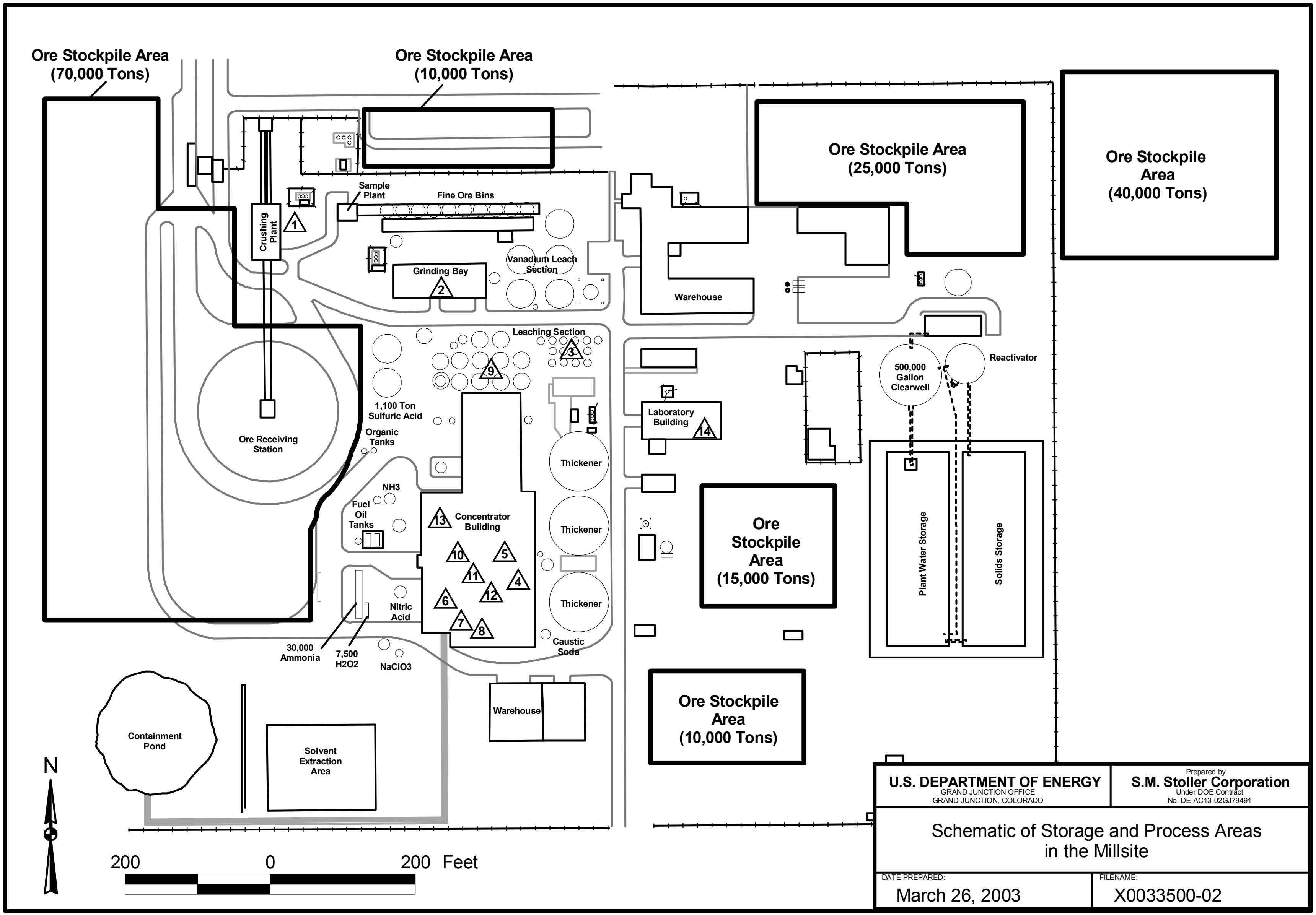

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Figure 3–3. Schematic of Ore Storage and Process Areas in the Millsite
(from 1973 Environmental Report and 1975 Safety Analysis Report)

According to the 1975 *Safety Analysis Report* (Dames & Moore 1975), at that time approximately 300,000 tons of ore were in an open stockpile covering an area of about 252,000 square feet (ft^2). Concerns about fugitive dust emissions from the stockpile prompted an evaluation of alternative methods for temporarily storing ore. However, it was concluded that the open stockpile was the preferred configuration, although provisions were made to wet the stockpile when meteorological conditions were severe enough to result in significant levels of fugitive dust (Dames & Moore 1975). It is not clear if this practice was actually implemented or at what frequency it would be required. Although the *Safety Analysis Report* refers to “an ore stockpile,” information from Atlas (1973) and aerial photos of the site during this period of operation indicate that ore was actually stored in multiple piles at various locations on the site (see Figure 3–2, Figure 3–4, and Figure 3–5). Merritt (1971) refers to a storage pad where ores were stored in various lots ranging from 50 to 1,000 tons. This pad is probably the ore receiving station where ores were segregated by type (Figure 3–3).

Mine-run ore was removed from the storage areas and fed through a series of crushers until it could pass through a ¾-inch screen. Some crushed ore was stored in stockpiles; it appears that the most finely crushed ore that was ready for further processing was stored in 10 fine-ore storage bins, five of which provided feed to the acid grinding circuit and four to the alkaline grinding circuit. The tenth bin was the reject bin; materials in this bin were returned to the stockpiles (Dames & Moore 1975; fine-ore bins are labeled on Figure 3–3).

Fine ores were ground and sent through the appropriate circuit for ore type, each of which included a series of steps involving extraction, filtration, and precipitation until the final product was recovered. Different circuits required the use of different chemicals for processing the ores and recovering the final concentrated product. Detailed descriptions of processing steps and chemistry are given in Merritt (1971), Atlas (1973), Dames & Moore (1975), and Shepherd Miller, Inc. (2001), among other reports. Water used in processing was drawn from the Colorado River and piped to a treatment tank (labeled “reactivator” on Figure 3–3) for pH adjustment and removal of hard water constituents (Dames & Moore 1975). Slimes and other precipitates in the treatment tank were removed from the tank as underflow and returned to the Colorado River (Merritt 1971). Clarified water overflowed to two settling ponds in series (labeled on Figure 3–3). The clean overflow from the solids settling pond flowed to the clean water settling pond. Calgon was added to clarified water from the clear water settling pond and then pumped to a large storage tank called the “clear well” for use as plant feed water. The sludge from the solids pond was periodically transferred to the tailings pile.

Process descriptions and flow sheets indicate that all processing was carried out using closed circuits; a significant portion of the liquids used were recycled and reused. Leached tailings from all process circuits were combined into a common sump (presumably located in the processing area) and pumped to the tailings disposal ponds via distribution pipes (Figure 3–4) located on three sides of the tailings pond (Merritt 1971). The slurry had a nearly neutral pH as a result of mixing materials from the alkaline and acid circuits. Decanted water from the tailings pond was removed through piping under the main pond to two small settling ponds (Drain Sumps on Figure 3–4) and barium chloride ponds ($BaCl_2$ Ponds on Figure 3–4) on the northeast and southeast sides of the tailings pond outside the tailings dike. Part of this water was recycled to the water treatment plant for eventual reuse in the mill circuit; the remainder flows through the ponds in series. A small amount of barium chloride ($BaCl_2$) was added to the second pond to coprecipitate radium from solution along with barium sulfate ($BaSO_4$.) Clarified water from the second pond was discharged to the Colorado River (probable location shown on aerial photo Figure 3–4—muddy “delta” area) (Merritt 1971).

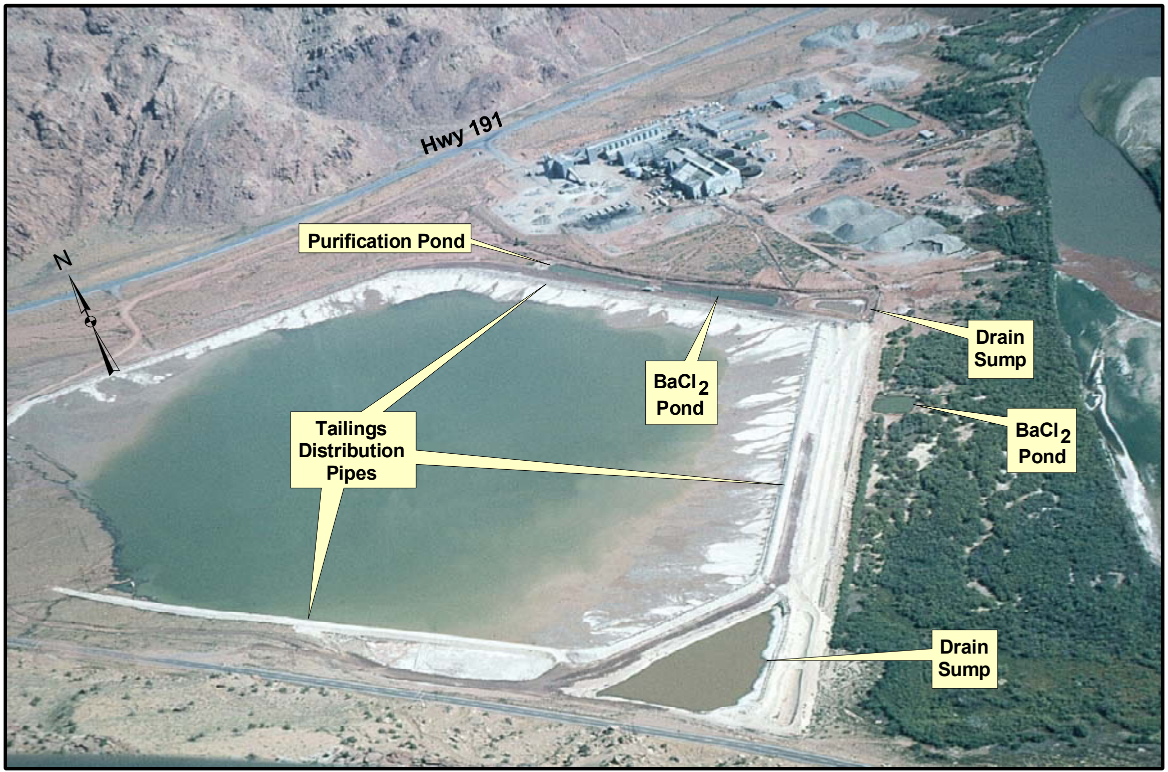

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Figure 3–4. Atlas Tailings Pile Photo (1966)

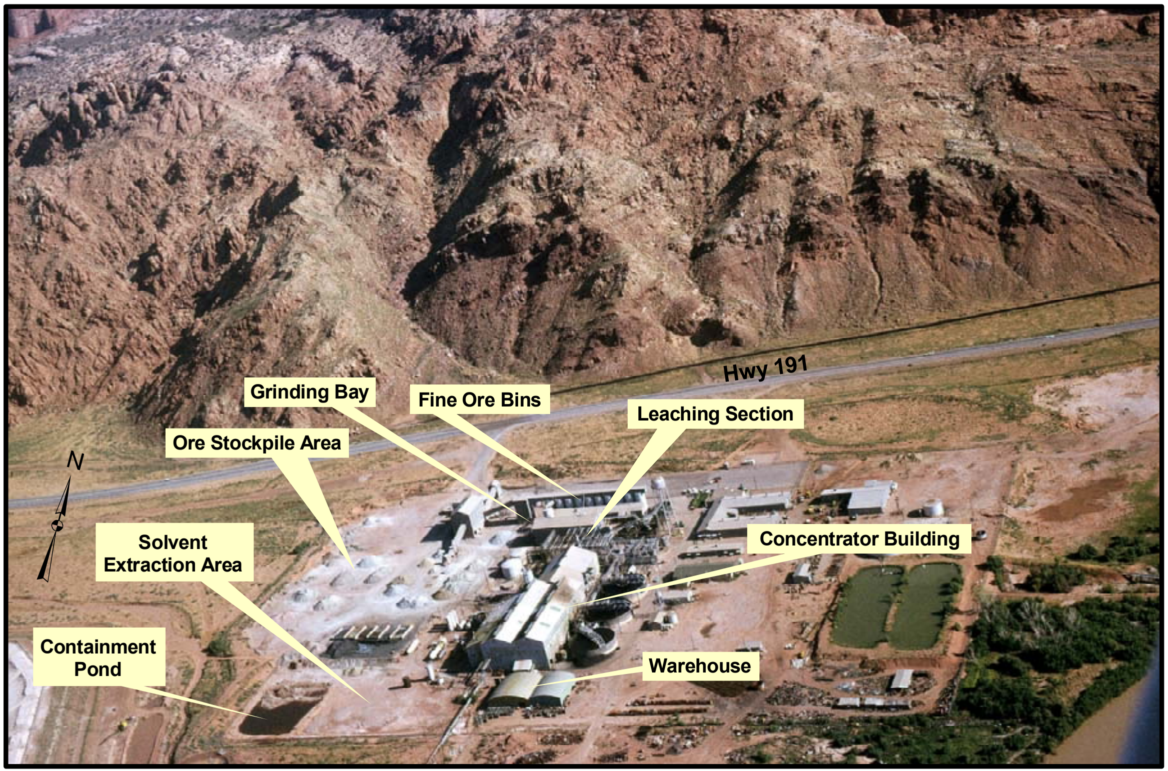

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Figure 3–5. Atlas Millsite Photo (1973)

3.2.2 Later Operations (1974−1984)

In about 1974, Atlas made several modifications to its ore processing operations. These changes were proposed in the 1973 Environmental Report (Atlas 1973) and were implemented shortly thereafter. The major modifications were:

- Construction of an acid-leach processing circuit to replace the one destroyed in a 1968 fire.
- Modification of the alkaline-leach circuit to reduce the volume of liquid effluents disposed of in the tailings pond.
- Elimination of direct discharge of effluent (liquids and solids) to the Colorado River.

A general schematic of the early process configuration and the proposed modifications (from Atlas 1973) is shown in Figure 3–6. Numbers in the upper corner of the boxes in this figure correspond to numbers (in triangles) on Figure 3–3, which are the locations where those process activities took place. Perhaps the most significant effect of the process modification was the decrease in water usage.

During the early processing period, water withdrawal from the Colorado River for use in processing was estimated at 1,300 gallons per minute (gpm). Implementation of process modifications aimed to decrease water usage an order of magnitude to 130 gpm (Dames & Moore 1974). With less effluent for disposal, evaporation of liquids from the tailings pond was adequate to keep pace with processing, and discharge to the Colorado River was no longer required. Presumably, use of the settling ponds adjacent to the southeast side of the pile was discontinued. Also, sludges from the initial treatment of water withdrawn from the Colorado River were disposed of in the tailings impoundment rather than discharged to the river. From 1982 to 1984, only an acid leach process was used (NRC 1999), and no process water was neutralized (Western Technologies, Inc. 1989). The 1989 ground water corrective action plan (Western Technologies, Inc. 1989) suggests that this resulted in disposal of low pH process water and increased metals mobilization. Atlas submitted a license renewal application in 1984 (Atlas 1984); however, at that time the site went on standby, and processing operations did not resume.

3.3 Chemical Inventories and Storage Areas

Several documents report on chemicals and quantities of chemicals that were used and stored at the site. An estimate of daily chemical usage from the 1973 Environmental Report (Atlas 1973) is listed in Table 3–1. It is not clear if the quantities are based on actual usage during early operations or on estimates for future usage after the proposed 1973 process modifications. However, this report does provide some idea of the types and quantities of chemicals that would have been expected at the site. The form of chemical storage and storage capacity is listed in Table 3–2 from the 1973 Environmental Report. Some of the storage locations, particularly storage tanks, are labeled on Figure 3–3. It is likely that some of the materials, particularly those stored in bags or used in small quantities, were stored in the laboratory, warehouse, or other general storage location.

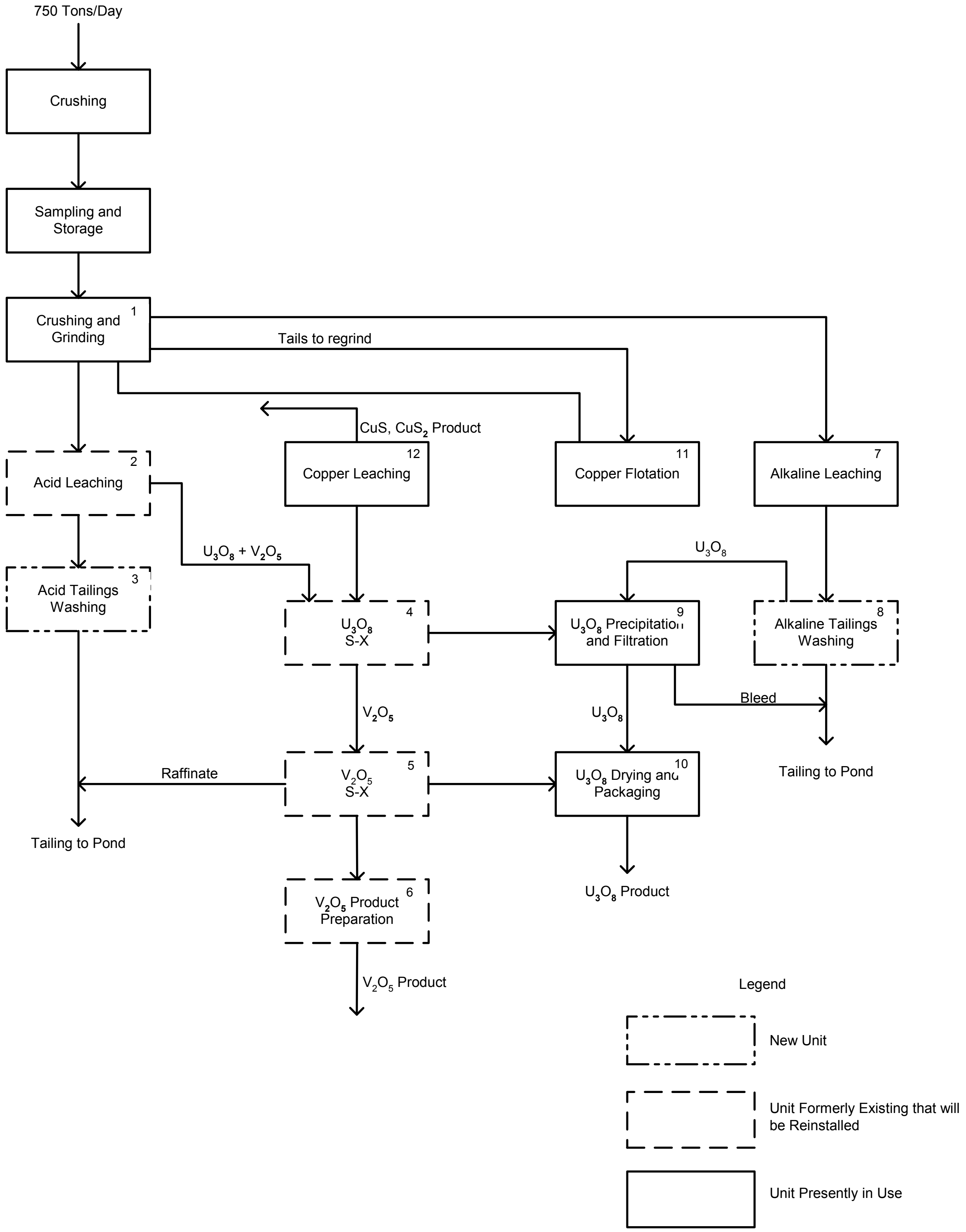

Figure 3–6. Process Circuits for Early 1970s and Proposed Modifications for Later Operations (from Atlas 1973 Environmental Report)

Table 3–1. Estimated Reagents Consumption—1973 Environmental Report (reported in pounds per day)

Reagents	Acid Circuit	Alkaline and Copper Circuit	Other	Total (pounds per day)
H_2SO_4, 95 wt %	200,000−300,000	300		200,000−300,000[a]
NaOH, 50 wt %		16,800		16,800
Anhydrous NH_3				10,000[b]
Na_2CO_3	8,000			8,000
$NaClO_3$	2,500		2,500	5,000
H_2O_2, 30 wt %		850		850
MnO_2 (native)		120		120
Iron powder	2,100	270		2,370[b]
Flocculant	75	38	2	115
Guargum	500	250		750
Potassium amyl xanthate		6		6
Frothing agent		2		2
Solvent extraction (organic)	100	1		101
Solvent extraction (kerosene)	530	70		600
$BaCl_2$			35−70	35−70
Aluminum sulfate			5	5

[a] Depends on the ore lime content.
[b] Estimated.

Table 3–2. Reagents Storage—1973 Environmental Report

Product	Storage Capacity	Product	Storage Capacity
Sulfuric acid	412,000 gallons	Hydrogen peroxide	7,500 gallons
Caustic soda (50% solution)	10,500 gallons	Ammonia	36,000 gallons
Anhydrous ammonia	69 tons	Propane	17,500 gallons
Sodium chlorate	30 tons	Grinding balls	40 tons
Burner fuel	51,500 gallons	Manganese (MnO_2)	Bag Storage
Diesel #2	4,000 gallons	Amine	Drum Storage
Gas	4,000 gallons	Isodecanol	Drum Storage
Soda ash	200 tons	Flocculant	Bag Storage
Hydrated lime	Bag Storage	Xanthate	Drum Storage
Powdered iron	Bag Storage	Aerofroth Frother	Drum Storage
Barium chloride	Bag Storage		

The following chemicals and water treatment compounds were received in drums or bags in small lot quantities:

Aluminum nitrate
Antifoam B
Antifoam HP
Isodecanol
Santosite, sodium sulfite
Soda, caustic flake
Sodium chlorate
Sodium hexametaphosphate
Starch
Tributyl phosphate
Urea fertilizer
Water treatment chemical
Xanthate

Estimated daily consumption of chemicals and reagents from NRC's 1979 EIS is presented in Table 3–3. Estimates of quantities of chemicals used are somewhat higher than earlier estimates, but types of chemicals used were largely the same. The 1975 *Safety Analysis Report* (Dames & Moore 1975) evaluated the types of site materials that could spill and enter the Colorado River. According to the report, possible contaminants would be of two types: chemicals and fuel or tailings materials. All the storage tanks were reported to be in one area of the mill and were said to be located within a bermed collecting area that had a capacity of more than one million gallons. No records have been found that mention any significant spills of chemicals or reagents.

Table 3–3. Estimated Daily Consumption of Chemicals and Reagents—1979 EIS (reported in pounds per day)

Reagents	Acid Circuit	Alkaline and Copper Circuit	Other	Total (pounds per day)
H_2SO_4, 95 wt %	240,000−360,000	730		241,000−360,000
NaOH, 50 wt %		40,100		40,100
Anhydrous NH_3[b]			15,000	15,000
Na_2CO_3	9,600			9,600
$NaClO_3$	3,000		3,000	6,000
H_2O_2, 30 wt %		2,100		2,100
MnO_2 (native)		290		290
Iron powder[b]	2,500	650		3,150
Flocculant	90	90	3	183
Guargum	600	580		1,180
Potassium amyl xanthate		15		15
Frothing agent		5		5
Solvent extraction (organic)	120	2		122
Solvent extraction (kerosene)	640	170		810
Aluminum sulfate			8	8

[a] Depends on the ore lime content.
[b] Estimated.

The 1973 Environmental Report (Atlas 1973) indicates that reclamation of the site will be required when milling operations cease and notes that a detailed reclamation plan will probably be prepared in the near future (1- to 2-year time frame).

The *Safety Analysis Report* (Dames & Moore 1975) has a two-page discussion about reclamation, including a discussion of absorptive capacity of soils between the tailings pile and the river. The report has geotechnical information on the pile and the site, presumably to support the reclamation approach (reclaim in place). The report also presents data for ground water monitoring between December 1972 and April 1974.

3.4 Potential Source Areas

A review of historical documents, drawings, and photographs indicates that several areas on the Moab site could have served as potential sources of ground water contamination at some point in time. Most of these features are shown on the schematic in Figure 3–3 or in the photos in Figure 3–4 and Figure 3–5. The most prominent of these potential sources are described in this section.

3.4.1 Tailings Pile Area

The pond on the tailings pile has undoubtedly served as the biggest source of ground water contamination, at least in the early years of operation of the mill (Figure 3–4). This is when the pond received the greatest volume and diversity of processing effluents; later many of the processing fluids were recycled. In addition, the base of the tailings pond probably had a higher permeability than in later years. As milling progressed and more tailings were disposed of, fines settled out in the center of the pile with coarser grains (sometimes referred to as the tailings beach because of the sand-like character) around the outside of the pond. The fine-grained material essentially served to seal off portions of the pond bottom along with precipitates from the pond fluids (Atlas 1973).

Besides the tailings fluids disposed of in the tailings pond, other materials have been buried in the tailings pile. Drums containing vanadium sludge are known to be disposed of in the western corner of the pile (Dames & Moore 1984, see Figure 3–7). Anecdotal evidence suggests that other materials could also be buried in the pile and have the potential to be leached to the ground water (NRC 1996b).

Figure 3–7. Tailings Pile Source Areas

Several features associated with the tailings pond could have served as discrete sources of contamination during their operation. These features include drain sumps, barium chloride ponds, a French drain system, purification pond, and infiltration gallery. All of these features contained or transported tailings-related fluids. Solids that accumulated in some of the ponds were excavated and consolidated with the tailings during site reclamation; other ponds may have just been backfilled (SMI 2001).

3.4.2 Millsite Area

Several potential source areas on the millsite area are shown in Figure 3–2, Figure 3–3, and Figure 3–5. This is the main area where uranium ore was stockpiled, including ores that had already been partially crushed. Precipitation seeping through these stockpiles could have leached some ore constituents. Tanks for process-related fluids and chemicals were also located in the millsite area. Any leaks or spills could have served as contaminant sources at the time. The solvent extraction area shown in the southwest corner of Figure 3–3 was associated with several emulsion ponds; a “containment” pond is also shown in this area. The nature of these ponds is not well known. Various drains, sumps, and piping were all used in the processing circuits in this area and represent potential contaminant sources. Interviews with site personnel indicated that fluids associated with resin regeneration were disposed of in a “catch pit” in the southeastern portion of the millsite area (ORNL 1998a). These fluids were said to be a nitric acid solution impregnated with uranium. Also in that area of the site were trash piles and trash disposal trenches. These can be seen in Figure 3–5 in the southeast corner of the photo. The trash disposal area also served as a disposal area for wood chips and other debris that were processed with the uranium ores. The wood chips were from timbers or other materials recovered with the ores. These materials were removed from process liquors by screens and disposed of at various places throughout the site (SMI 2001).

3.5 Water Usage Rates and Pond Levels

3.5.1 Water-Use Budget

3.5.1.1 Early Process Water Budget

A summary of water used during the 1971−1972 period is presented in Table 3–4. The average flow of water entering the plant and exiting from the purification pond during this period is approximately 1,228 gpm and 1,040 gpm, respectively. The average difference between the two flows during this time is 188 gpm. This is the average volume of liquid that flowed into the tailings pond resulting from ore processing.

The tailings slurry contained approximately 11.5 percent tailings by weight (Dames & Moore 1973). Because the weight of water is 8.34 pounds per gallon (lb/gal), and by difference, it comprises 88.5 percent of the slurry, the tailings slurry had a weight of 9.42 lb/gal. The difference of 1.08 lb/gal is the weight of the tailings in the slurry. Records indicate that the mill was processing the ore at the rate of approximately 1,000 tons per day. Consequently, the average intake to the plant of 1,228 gpm was required to move the slurry to the tailings pond. The average excess of 1,040 gpm that returned to the Colorado River originated as the supernatant from which the tailings had settled out. The average difference of 188 gpm was the fraction that remained on the tailings pile. Part of this fraction evaporated from the pile, and the remainder seeped into the ground.

Table 3–4. Representative Water Budget for the Atlas Millsite During the Early Processing Period

Year	Month	Average Monthly Intake to Plant (gpm)	Average Monthly Discharge from Purification Pond (gpm)	Difference (gpm)
1971	January	1,454	1,321	133
	February	1,540	1,320	220
	March	1,405	1,205	200
	April	1,565	1,169	396
	May	1,323	1,109	214
	June	1,858	1,387	471
	July	380	0	380
	August	1,324	552	772
	September	1,327	1,061	266
	October	1,093	1,045	48
	November	1,489	1,335	154
	December	1,617	1,397	220
1972	January	1,653	1,432	221
	February	1,634	1,395	239
	March	1,299	1,542	−243
	April	935	837	98
	May	961	751	210
	June	980	874	106
	July	780	368	412
	August	1,183	886	297
	September	1,169	965	204
	October	1,031	1,120	−89
	November	1,086	1,137	−51
	December	1,384	1,385	−1
1973	January	1,466	1,446	0
	Averages	1,228	1,040	188

Source: Dames & Moore (1973), Page B-3.

Dames & Moore (1974) estimates that the average seepage rate from the tailings pond was about 75 gpm during the 1971−1972 period. This estimate is based on a water balance calculation and a corollary estimate that the ponded surface area of 45 acres had a unit seepage-loss value of 1.6 gpm per acre of pond.

3.5.1.2 Later Process Water Budget

Atlas Minerals modified its milling process after 1974. Details of the modified milling process are described in the 1973 Environmental Report (Atlas 1973). A key element of the revised milling process is that water use drops to approximately 130 gpm as a result of a more efficient milling procedure and recycling of the water that previously flowed back to the Colorado River from the tailings pile.

As a result of the modified milling process, the tailings pond was projected to shrink to an area of approximately 28.3 acres, and seepage rate was forecasted to drop (Dames & Moore 1975). Process changes implemented in 1974 also altered the liquids-to-solids ratio of effluent entering the tailings pond. The new ratio became 2.7, consisting of 521.6 gpm (261,000 lb/hour) liquid and 98,000 lb/hour solids. The difference between inflow and outflow from the pond (121.6 gpm) was projected to evaporate entirely; however, the 28.3-acre liquid pool must have been the source of some seepage as well. The amount of that seepage, assuming that it was proportional to the pond area and that the seepage rate remained constant at 1.6 gpm/acre, would have been approximately 45 gpm.

3.5.2 Pond Levels

The tailings pond began with an original "starter dike" whose crest was at an elevation of 3,974.7 ft; the original ground elevation adjacent to the starter dike was approximately 3,968.2 ft (Dames & Moore 1973, Plate 3A). The starter dikes were extended, and supplemental tailings dikes were constructed to increase the height of the embankment system. The embankments were raised by crawler tractor pushing the coarse tailings fraction up from the pond. Merritt (1971) states that the embankments were raised approximately every 3 years to create more storage capacity for tailings. A major expansion of the original starter dike system began in late 1967 (Dames & Moore 1973) and again in 1974 (Dames & Moore 1975).

The water surface elevation of the tailings pond was documented for only two dates: April 12, 1971, and December 11, 1972, when the respective water surface elevations were surveyed to be 4,022.2 ft and 4,022.6 ft (Dames & Moore 1975, and Dames & Moore 1973, Plate 3A). As an operating practice, the maximum depth of the tailings pond was maintained below 5 ft (Dames & Moore 1975). Shepherd Miller, Inc. (SMI 2001) reported that the final elevation of the tailings embankment was 4,058 ft above mean sea level.

ORNL (1998) compared water elevation data collected for the pond on the tailings pile between 1989 and 1994 with river elevations during the same time span. No correlation was discerned between the two sets of data. Therefore pond levels were not likely controlled by river flow rates.

3.6 Atlas Decommissioning and Reclamation Studies

3.6.1 Tailings and Surface Reclamation

Decommissioning and reclamation plans, though very general, were routinely submitted to NRC by Atlas as part of the documentation required for licensing. For example, the 1973 Environmental Report (Atlas 1973) indicated the pile would be covered in place to a depth sufficient to meet radiation standards. Radiation surveys and restrictions on future use of the site were planned. However, during the period of operation, most of the engineering and geotechnical studies of the pile focused on modifying the embankment system to increase the capacity of the tailings disposal system.

NRC's 1979 Final EIS for operation of the Moab mill evaluated several alternatives for final tailings disposition (NRC 1979). Several on-site alternatives were evaluated, each with a different cover design. Off-site disposal at two alternative sites was also considered. The preferred alternative identified at the time was on-site stabilization with the use of a clay cover, which would be vegetated with appropriate plant species. It was noted that reclamation would begin as soon as the area achieved sufficient dryness.

Atlas' original reclamation plan was prepared in 1981 (Dames & Moore 1981) and described in Atlas' 1984 license renewal application (Atlas 1984). A revised plan was prepared in August 1988 (Canonie 1988) with several modifications to the existing plan, including (1) the top of the impoundment was revised to have a depressed channel instead of a domed top to promote southwest drainage and reduce settlement/cover damage; (2) the thickness of the radon barrier was reduced to meet revised requirements for radon emissions; (3) riprap/rock armor erosion protection was added; and (4) a plan was presented to relocate and reconfigure Moab Wash to minimize flood impact (Canonie 1988).

Interim cover placement on the pile began in August 1989 (NRC 1999) and was completed in phases as the center of the pile dried up. Placement of the cover was finished in November 1995 (NRC 1999). Photos of the site before and after interim cover placement are shown in Figure 3–8 and Figure 3–9, respectively. Many of the on-site buildings and equipment were dismantled and deposited in the tailings pile. Reports indicate that much of this material was disposed of in the southern portion of the pile (SMI 2001).

In July 1990, Atlas began pumping water from wells in the pile (shown as the PW series on Plate 1) to the top of the pile for evaporation to accelerate dewatering and consolidation of the pile (Canonie 1994). They initially used 10 extractions wells, but the number of wells was subsequently reduced to seven. Estimated pumping rates were between 131,000 and 145,000 gallons per month (Canonie 1994). The 1996 photo (Figure 3–9) indicates that pumping ceased before that time.

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Figure 3–8. Before Interim Cover Placement (1984)

Figure 3–9. After Interim Cover Placement (1996)

During the time that the interim cover placement and pile dewatering was occurring, Atlas was attempting to obtain NRC approval for final reclamation plans. It appears that there were numerous iterations of NRC reviews and subsequent reclamation plan revisions. The 1988 reclamation plan (Canonie 1988) was revised (Canonie 1992), and an environmental report supplement was prepared to satisfy NEPA requirements (HLA 1993). In 1993, NRC published an Environmental Assessment and Finding of No Significant Impact (FONSI) amending the Atlas license and approving the reclamation plan. The FONSI and license amendment were subsequently withdrawn due to technical issues raised by the public during review of the FONSI and environmental report supplement (NRC 1999). NRC later made the decision to prepare an EIS for reclamation of the site and do a more detailed analysis of the major issues raised regarding the proposed alternative (on-site disposal). The Notice of Intent to prepare an EIS was published in the *Federal Register* on March 30, 1994 (NRC 1999). At that time investigators began to recognize the importance of ground water as a component of the reclamation plan, and many of the subsequent studies focused on characterizing the ground water, the interactions of the pile and ground water, and interactions between the ground water and surface water.

3.6.2 Ground Water/Surface Water

This section summarizes the objectives and major conclusions of historical studies that focused on ground water and surface water at the Moab site. Specific study results pertinent to this SOWP and development of the site conceptual model are presented in Section 3.8. Most of the pre-1996 studies focused on metals and radionuclide contamination rather than potential effects of the tailings pile on the ground water and surface water.

In 1989, Western Technologies, Inc. (1989) prepared a ground water Corrective Action Plan for Atlas. The plan proposed minimal corrective action and called for natural dewatering and flushing to attain proposed alternate concentration limits in ground water. The minimal action was justified on the basis of the ability to maintain institutional controls at the site, the naturally poor water quality that existed in the alluvial aquifer before the Atlas mill was constructed, and the dilution effect of ground water discharging to the Colorado River (thereby reducing any potential adverse impacts). It was assumed that the Title II ground water standards to be met were those stipulated by NRC as part of Atlas' license (see Table 3–5). The plan also mentioned EPA's proposed Title I UMTRA ground water regulations, which permitted a 100-year period for natural flushing to attain standards. This alternative was proposed for the ground water associated with the Atlas site; it was estimated that 75 years would be required for natural flushing to meet applicable standards (Western Technologies, Inc. 1989). As noted in Section 3.6.1, dewatering of the pile began in July 1990.

Table 3–5. NRC-Stipulated Water Quality Standards—Atlas Moab Mill[a]

Constituent	Concentration
Chromium (mg/L)	0.08
Gross Alpha (pCi/L)	33
Molybdenum (mg/L)	0.05
Nickel (mg/L)	0.06
Radium-226 and -228 (pCi/L)	5
Selenium (mg/L)	0.01
Uranium (mg/L)	0.006 (4.0 pCi/L)
Vanadium (mg/L)	0.04

[a]Stipulated by NRC in a letter to Atlas Minerals dated December 14, 1988.

After its review of the 1989 Corrective Action Plan, NRC requested that additional information be provided to support the conclusion that natural flushing would be an effective corrective action for ground water. In response, Canonie Environmental (1994) prepared a report for Atlas stating that estimated seepage rates from the pile were about 33 gpm. On the basis of this analysis, Canonie concluded that the pile would take about 56 years to drain. It appeared that effects of pile drainage on surface water quality were not considered to be significant.

While NRC's preparation of its draft EIS for reclamation of the mill was in progress, many technical studies and reports were prepared in support of the EIS as a result of comments received on the earlier Environmental Assessment. NRC received extensive comments on the Environmental Assessment from the USF&WS, who identified the need for additional studies before the reclamation project could be approved. Of concern were threatened and endangered fish in the river and the potential impacts to critical fish habitat located adjacent to the millsite.

As a result, NRC (in an October 1994 letter) requested additional river and sediment sampling from Atlas. Atlas submitted additional data in January 1995 indicating that water and sediment data downstream of the Atlas site were not appreciably different from data collected upstream of the site. Reviewing agencies did not feel that the data collection and evaluation were adequate and requested that additional sampling be performed (WestWater Engineering 1995).

Subsequently, a number of government agencies participated in an on-site examination of the Atlas site on April 5, 1995, to determine appropriate sampling locations and protocols. Participants included the National Park Service (NPS), USF&WS, Utah Department of Wildlife Resources, Oak Ridge National Laboratory (ORNL), NRC, and the U.S. Geological Survey (USGS). On the basis of input from the participating agencies, NRC prepared a sampling and analysis plan (Federal Plan) to better determine the effect of the millsite, if any, on the biota in the area and on the endangered fish habitats.

A modified sampling and analysis plan was implemented and results reported in July 1995 (WestWater Engineering 1995); water, biota, and sediment were analyzed. Constituents analyzed consisted of selected metals and radionuclides. Ammonia was not identified as an issue at that time. The general conclusions of the study were that contaminants in upstream and downstream sediments and biota were not significantly different and that negative impacts to endangered fish in the vicinity of the mill site were considered unlikely.

Using the results of the 1995 study (WestWater Engineering 1995), NRC completed a biological assessment for inclusion in its draft EIS for reclamation of the Atlas site. The assessment indicated that the proposed action was unlikely to adversely affect threatened and endangered species in the site vicinity (Appendix B in NRC 1999). It was noted that concentrations of ammonia and several other constituents occasionally exceeded state water quality standards for protection of aquatic life, but that this likely occurred only under low-flow conditions of the river. The assessment stated that only a small mixing zone near the east bank of the river was affected by ground water discharge and that the proposed tailings pile stabilization would reduce these hazards. Impacts of reclamation activities, such as siltation, were evaluated and mitigative measures identified to minimize these impacts. The conclusion was reached that reclamation activities should not adversely affect threatened or endangered fish at the population level; potential effects on individuals were inconclusive.

In about 1996, investigators began to direct their attention to ammonia contamination at the site and found that it was directly related to the presence of the tailings pile. At that time more detailed studies and specific evaluations were performed in response to requests by NRC for information to support Atlas' proposed site reclamation and license amendment.

To better understand the distribution of ammonia in the Colorado River and its relationship to the Moab tailings pile, the State of Utah conducted an intensive ammonia sampling effort in September 1996 and January 1997. The results of these sampling events were transmitted to NRC in letters dated December 1996 and June 1997, respectively. The State concluded that the tailings pile was the major source of ammonia contamination and that the mixing zone for ammonia along the river could be up to a mile long. They indicated that further studies were needed to evaluate the potential presence and distribution of other contaminants that could be coming from the site and that the pathways through which ammonia was moving from the pile to the ground water and subsequently discharging to the river needed to be better understood. Their June 1997 letter to NRC recommended that additional studies be completed and issues resolved during NRC's planned review of the Atlas ground water Corrective Action Plan.

During preparation of its draft EIS, NRC initiated consultation with the USF&WS. The USF&WS indicated that it would issue a biological opinion on the proposed project after review of NRC's biological assessment. After reviewing the initial biological assessment, the USF&WS stated that it "could not conclude that the tailings pile, if capped in place, would not harm the endangered fish of the Colorado River system, or would not result in the destruction or adverse modification of designated critical habitat." The USF&WS recommended collecting additional data to fill data gaps. In response, NRC completed additional review of data on water quality and biota and an undated evaluation of potential impacts to threatened and endangered species (Appendix B in NRC 1999). Results were submitted in a biological assessment supplement completed by NRC in January 1997 and submitted to USF&WS for their review. Major conclusions of the supplement were that, except for ammonia, the pile contributes only small amounts of other contaminants to the Colorado River. Although some elevated levels of selenium and mercury were present in fathead minnows collected from the river near the pile, these constituents did not appear to be site-related. NRC indicated that if ammonia concentrations persisted in the future, the need for additional mitigative measures would be considered.

The USF&WS issued a draft biological opinion in June 1997. Because the draft opinion expressed concerns that had not been addressed, the parties involved agreed to conduct further studies. In September 1997, Oak Ridge National Laboratory's Grand Junction, Colorado, office (ORNL-GJ) began planning to conduct additional sampling. Several organizations took place in this planning, including the NPS, USF&WS, State of Utah, Grand Canyon Trust, and ORNL-GJ. The biggest concern was the discharge of ground water that was producing elevated levels of ammonia in the river.

ORNL-GJ conducted the ground water study in November and December 1997 and submitted a report in January 1998 (ORNL 1998a). Aside from refining estimates of important hydrologic parameters, the report concluded that except during very high river stages, the base of the tailings pile was above the water table and not in direct contact with ground water. The contaminant plumes from the pile were interpreted to be mature, that is they had probably been discharging to the river for a number of years. A potential source of the uranium ground water plume was identified as a "catch pit" near well TP-02, which, according to interviews with Atlas personnel, received nitric acid solution impregnated with uranium. ORNL-GJ analysis indicated that a 2.8 mg/L average concentration of uranium in ground water downgradient of the pile would be expected to persist indefinitely.

After reviewing this report, the USF&WS said that additional modeling was needed to determine the long-term impacts of the pile on the river. ORNL-GJ completed a supplemental ground water modeling study and submitted a modeling report in February 1998 along with a model of pile seepage (ORNL 1998b, 1998c). The focus of these studies was to further refine the understanding of impacts of pile seepage to the ground water and, in turn, to the surface water. The reports concluded that the bulk of pile drainage was expected to occur within the first 100 years and that contaminant concentrations at the river would continue to increase for 9 more years. On the basis of modeling that assumed source removal, ground water was predicted to return to pre-1956 water quality levels after at least 35 years.

A surface water study (HLA 1998) was being completed in tandem with ORNL-GJ's ground water study. The Colorado River was sampled along transects at distances of 10 ft, 25 ft, and 50 ft from shore. Samples were collected from three depths at each location. Ground water samples were also collected from three wells immediately adjacent to the river. All samples were analyzed for ammonia, molybdenum, and uranium. Major findings of the study were that ground water discharging to the river appeared to be diluted from factors of about 300 to almost 800. Sampling results showed that a mixing zone for ammonia (exceeding state standards) extended approximately 4,000 ft downstream of the site and 25 ft transversely into the channel; the uranium mixing zone extended 12,000 ft downstream and 50 ft into the river. Results indicated that uranium and molybdenum concentrations in the river were elevated above ambient levels, but that maximum concentrations were well below lowest chronic exposure benchmarks for aquatic organisms.

The USF&WS submitted a final biological opinion to NRC in July 1998 saying that a ground water corrective action plan had to be developed that was related to the Reclamation Plan. The biological opinion stated that, as proposed, the reclamation project would jeopardize the razorback sucker and Colorado pikeminnow; concerns were over water depletion as well as destruction or adverse modification (chemical and physical) of critical habitat. They also questioned the validity of previous studies regarding pile seepage and biota characterization and indicated that the effects on endangered fish of ground water discharging to the river could be significant. The USF&WS developed alternatives to avoid the likelihood of jeopardy and habitat destruction or modification.

The major elements of a reasonable and prudent alternative were:

- Ground water corrective action—dewater tailings pile, cleanup ground water to meet surface water standards in 7 years, remove jeopardy to listed fish in 10 years, monitor surface water quality.
- Incorporate ammonia standards in Atlas' license.
- Conduct bioassay studies to evaluate toxicity of ammonia plume on endangered fish and develop a site-specific ammonia standard.
- Establish an ACL for protection of human health and aquatic life that would be met at a point of compliance.
- Provide a water depletion payment for the endangered fish recovery program.

In early 1998, before the final biological opinion had actually been issued, the USF&WS requested that USGS provide research and technical assistance to determine the potential impacts of the tailings pile to the endangered fish in the Colorado River to partially address requirements specified in the USF&WS final biological opinion. Atlas did make the water depletion payment: however, before the additional USF&WS requirements specified in the final biological opinion could be met, Atlas filed for bankruptcy and was relieved of its responsibility for site reclamation. Additional studies were performed in 2000 and results of all studies were published in December 2002 (Fairchild et al. 2002). Results of these are summarized in Section 3.9.4.

3.7 Post-Atlas Reclamation Studies/Developments

Atlas filed for bankruptcy in September 1998 and subsequently signed a Settlement Agreement with NRC, the State of Utah, and other creditors. According to this agreement, Atlas terminated its license in late 1999 and transferred certain assets into a Reclamation Trust. PricewaterhouseCoopers was named as fiduciary trustee of the Reclamation Trust and licensee for the site.

Although Atlas had already filed for bankruptcy, NRC issued its final EIS (FEIS) on the Atlas reclamation plan in March 1999 and the final Technical Evaluation Report in April 1999. The conclusions reached in the FEIS were that the Atlas-proposed on-site reclamation would significantly reduce impacts of contaminants entering the river but indicated that approval of the proposed reclamation would require that the reasonable and prudent alternatives and measures specified in the final biological opinion be met. It was noted that this would require additional data and analysis. All other environmental aspects of the proposed alternative were deemed acceptable.

The FEIS contained little evaluation of the ground water and resulting impacts of the proposed reclamation of the tailings pile. Presumably, NRC was deferring this aspect of site reclamation to the recommended additional studies. Public comments on the proposed alternative along with responses to them were included in the FEIS. A record of decision for the FEIS was never published.

3.7.1 Trustee Studies

While activities initiated before the Atlas bankruptcy were being completed, PricewaterhouseCoopers, the trustee for the site, was proceeding to satisfy other requirements of the final biological opinion. Studies were undertaken to characterize the fluids remaining in the pile in order to design an appropriate dewatering system (SRK 2000) and better understand pile consolidation. Hydrogeologic and geochemical studies of the site ground water and interactions with the Colorado River were also initiated to assist in evaluating alternatives for ground water corrective action (SMI 2001). Most of the fieldwork was performed in summer 2000, and a report was published in April 2001. Site conditions in 2000 were similar to those at the site in 2001 (shown in Figure 3–10). The geochemical studies included characterizing ammonia in the Colorado River adjacent to the site (provided in interim report 2001; Appendix B-6 of SMI 2001).

Geotechnical and geochemical studies of the tailings pile (SRK 2000) concluded that dewatering of the pile would decrease seepage rates to ground water to an insignificant amount and would be required before final cap placement. A vacuum-assisted wick drainage system was recommended as the dewatering option. The ground water and surface water studies (SMI 2001) focused primarily on ammonia and on determining the extent that site-related contamination affected water quality of the Colorado River. Results showed that only limited backwater and nearshore areas had ammonia concentrations that exceeded EPA chronic criteria for aquatic species with early life stages present. The maximum area in which ammonia levels were above EPA criteria during a below-average flow year (such as in May/June 2000 when the study was performed) was estimated as less than 0.5 mile long and no more than 35 ft from shore. The entire area was estimated at less than 3 acres. It was also noted that the EPA criteria for ammonia are based on safe levels for a 30-day exposure period; endangered fish in the river are more likely to be exposed for shorter periods, and the higher concentrations could still be considered protective.

Figure 3–10. Site Conditions in 2001

Comparisons of monitored ground water elevations with Colorado River elevation data collected by SMI (2001) during 2000 showed ground water at the Moab site discharged to the river during most of the year. Under baseflow conditions (river flows ranging from 3,000 to 4,000 cfs) the alluvial aquifer discharges to the Colorado River. Flow reversal, with the river feeding water to the aquifer in the form of bank storage, occurred only between the beginning of April and the first week in May. The flow reversal first began once river discharge climbed to 10,000 cfs, and the river subsequently reached a peak discharge of 17,000 cfs. Following the high runoff season, flow from the aquifer to the river resumed and persisted during the rest of the calendar year.

An additional study at the Moab site by SMI (2001) examined the potential for relatively low-TDS river water to recharge the alluvium and subsequently return water containing high ammonia levels to the river. Extensive monitoring during this investigation found no evidence of increased ammonia concentrations in either ground water or the Colorado River.

The investigation of ground water/surface water interaction concluded that removal of tailings would have no discernible impact on the loading of constituents to the river through ground water discharge, and that active ground water remediation would be required for 35 to 50 years to decrease ground water contaminant concentrations to levels that are protective of aquatic life in the river (SMI 2001). It was estimated that ammonia loading to the river in backwater areas at a rate of about 10 pounds per day or less would be protective of aquatic species. Different ground water cleanup methods were evaluated, including typical "pump-and-treat" systems and more passive barrier systems. It was concluded that the fastest restoration could be achieved by installing extraction wells to pump ground water from the aquifer (treatment of extracted ground water would follow); this alternative was also the most expensive of those considered.

3.8 Summary of Previous Documents/Studies

3.8.1 Documents Related to Site Operations

- Merritt (1971)—operations description
- 1973 Environmental Report (Atlas) (and two 1975 supplements)
- Safety Analysis Report (Dames & Moore 1975)
- Environmental Impact Statement (NRC 1979)
- Atlas 1984 license renewal application
- NRC Environmental Assessment for license renewal followed by a FONSI approving continued operation (1988)

3.8.2 Documents Related Mainly to Reclamation/Cleanup Plans

3.8.2.1 Tailings Pile

- Dames & Moore 1981—Conceptual Design and Cost Estimate—Tailings Pile Reclamation
- Canonie 1988—Reclamation Plan Uranium Mill Tailings Disposal Area
- Canonie Environmental 1992 —Atlas Corporation Reclamation Plan, Uranium Mill and Tailings Disposal Area
- HLA 1993—Environmental Report Supplement
- Smith Technology Corp. 1996—Final Reclamation Plan, Atlas Corporation Uranium Mill Tailings Disposal Area
- Senes Consultants Limited 1995—Screening Risk Assessment for Reclamation of Uranium Mill Tailings at Moab, Utah
- Woodward-Clyde 1996—Evaluation of Potential Seismic and Salt Dissolution Hazards at the Atlas Uranium Mill Tailings Site, Moab, Utah
- ORNL 1998c—Tailings Pile Seepage Model of the Atlas Corporation Moab Mill, Moab, Utah
- NRC 1996—Draft Technical Evaluation Report for the Proposed Revised Reclamation Plan for the Atlas Corporation Moab Mill (plus Final 1997 and April 1999 supplement)
- NRC 1998—Infiltration, Seepage, and Ground Water Contamination Modeling for the Atlas Corporation Uranium Mill Tailings Site; prepared by Center For Nuclear Waste Regulatory Analysis
- NRC 1999—Final Environmental Impact Statement Related to Reclamation of the Uranium Mill Tailings at the Atlas Site, Moab, Utah
- SRK 2000—Dewatering Design Plan
- SRK 2001—Tailings Geochemistry, Atlas Mill Site, Moab, Utah

3.8.2.2 Ground Water

- Dames & Moore 1975—Safety Analysis Report: monitoring requirements for early to mid-1970s
- Regular Atlas monitoring reports submitted to NRC
- EnecoTech Inc. 1988—Ground Water Hydrology Detection Monitoring Program
- Western Technologies 1988—Atlas Moab Mill Ground Water Detection Monitoring Program
- Western Technologies 1989—Atlas/Moab Uranium Mill and Tailings Corrective Action Plan
- Canonie 1994—Atlas Corporation Ground Water Corrective Action Plan; Uranium Mill and Tailings Disposal Area
- ORNL 1998a—Limited Ground Water Investigation of the Atlas Corporation Moab Mill, Moab, Utah
- ORNL 1998b— Tailings Pile Seepage Model of the Atlas Corporation Moab Mill, Moab, Utah
- ORNL 1998c—Supplemental Modeling and Analysis Report; Atlas Corporation Moab Mill, Moab, Utah
- Shepherd Miller, Inc. 2001—Site Hydrogeologic and Geochemical Characterization and Alternatives Assessment for the Moab Uranium Mill Tailings Site, Moab, Utah

3.8.2.3 Surface Water

- U.S. Department of Health, Education, and Welfare 1961. Stream surveys in the vicinity of Moab millsite—has surface water data
- Regular Atlas monitoring reports submitted to NRC
- WestWater Engineering 1995—Atlas Corporation Moab Mill Site Colorado River Sampling and Literature Review
- State of Utah intensive ammonia sampling—results transmitted via letters dated December 1996, June 1997
- Harding Lawson Associates 1998—Colorado River Sampling Report, Atlas Uranium Mill and Tailings Site, Moab, Utah
- Shepherd Miller, Inc. 2001—Interim Report: Ammonia Concentrations in the Colorado River Adjacent to the Atlas Mill Tailings, Moab, Utah
- Shepherd Miller, Inc. 2001—Site Hydrogeologic and Geochemical Characterization and Alternatives Assessment for the Moab Uranium Mill Tailings Site, Moab, Utah

3.8.3 Documents Related to Ecological Issues

- WestWater Engineering 1995—Atlas Corporation Moab Mill Site Colorado River Sampling and Literature Review
- NRC Biological Assessment and Supplement—Included in Appendix C of NRC 1999 FEIS
- USF&WS Final Biological Opinion 1998—Included as Appendix C of NRC 1999 FEIS along with related correspondence

- USGS 1999—Determination of a Safe Level of Ammonia that is Protective of Juvenile Colorado Pikeminnow in the Upper Colorado River, Utah
- Fairchild et al. 2002—A Site-Specific Assessment of the Risk of Ammonia to Endangered Colorado Pikeminnow and Razorback Sucker Populations in the Upper Colorado River Adjacent to the Atlas Mill Tailings Pile, Moab, Utah

3.9 Summary of Results from Previous Investigations

A technical information request for NRC prepared in 1994 (Canonie 1994) provides a good synopsis of the monitoring history of the Moab site. According to this report, ground water quality sampling of monitor wells began in 1979; sampling of surface water and tailings pile water began in 1981 and 1990, respectively. The report notes that a substantial amount of monitoring data is available, but that sampling protocols, analytical methods, and sampling locations changed many times over the years. Some locations were sampled for only brief periods, and the analytes, sampling and analysis methods, and detection limits varied over time. Therefore, a comprehensive evaluation of changes in ground water quality over time is somewhat complicated.

3.9.1 Tailings

3.9.1.1 Pond Chemistry

During the early and later milling periods the primary constituents being monitored for water quality were radium-226, thorium-230, radon-222, and uranium. Atlas Minerals monitored these constituents at the point where water entered the purification ponds and where the water was being discharged from the purification ponds to the Colorado River (Figure 3–4).

Radiochemistry results and major ion chemistry of the tailings pond liquid are reported in Atlas' Environmental Report (Atlas 1973) and summarized in Table 3–6. Results from the 1973 Atlas report are assumed to be indicative of pond chemistry during the later milling period. Results indicate that the dissolved solids (TDS) concentration, which was dominantly composed of sulfate, was approximately 150,000 mg/L during the milling period.

Table 3–6. Tailings Pond Chemistry for Milling Period

Constituent	Concentration
Radium-226	1×10^{-7} µCi/mL 100 pCi/L
Thorium-230	0.05×10^{-6} µCi/mL 50 pCi/L
Chloride	0.3 g/L (300 mg/L)
Sulfate	100 g/L (100,000 mg/L)
Arsenic	0.007 g/L (7 mg/L)
Total dissolved solids	150 g/L (150,000 mg/L)
Uranium	0.002 g/L (2 mg/L)
Vanadium	0.3 g/L (300 mg/L)

NRC collected two filtered samples of the tailings pond water in July 1987 (NRC 1996a), which is a few years after the mill had ceased operations in 1984, but prior to placement of the interim cover. Results of selected analytes are presented in Table 3–7.

Table 3–7. Tailings Pond Chemistry After the Milling Operations

Constituent	Concentration (mg/L)	
	Sample 1	Sample 2
Ammonia	2,150	2,400
Sulfate	26,000	30,000
Uranium	4.0	8.9
Total Dissolved Solids	22,800	23,900

3.9.1.2 Pore Fluid Chemistry

There were no pore-fluid chemistry analyses performed during the period of milling (1956−1984). However, Atlas analyzed the tailings pond fluid at least once during that period and presented the results in the 1973 Environmental Report (Atlas 1973). These results are summarized in Table 3–6. Although the results are from a pond sample and not from a pore-fluid sample, the 150,000 mg/L TDS provides an indication of the high salinity that might have been present in the pore fluid during milling.

Monitoring of pore fluid in the tailings pile began in 1990. Pore fluid samples from dewatering wells were collected semiannually and analyzed for the same constituents as ground water (see Canonie 1994 for further details). The duration of pore fluid monitoring is not known. A summary of TDS concentrations in water samples from the dewatering system is summarized in Table 3–8 (NRC 1999).

Table 3–8. TDS Concentrations From Tailings Dewatering Wells

Year	Total Dissolved Solids (mg/L)
1990	24,700
1991	25,065
1992	30,250

During the post-milling period, both ORNL-GJ (ORNL 1998b) and Steffan, Robertson, and Kirsten (SRK 2000) analyzed the tailings pore fluids. A summary of these analyses is presented in Table 3–9 for selected analytes. Note that the full analyte list in the SRK study is much more extensive.

Table 3–9. Summary of Tailings Pore Fluid Chemistry

Reference	Basis of Data	Undifferentiated (mg/L)		
			Range	Average
ORNL 1998b Table 3−9	Four pore fluid samples collected in November 1997	Ammonium-N Sulfate Uranium	1,070–3,940 15,786−31,484 19.8−26.5	2,320 22,363 23.5
SRK 2000 Table 5−10	Six pore fluid samples collected in May 2000	Ammonia-N Sulfate Uranium	38.2−3,430 2,726−26,322 2.44−55.3	1,414 13,713 22

3.9.1.3 Seepage Rates

Dames & Moore (1973 and 1974) estimated seepage into the tailings pile using a water budget analysis that was prepared as part of the design to expand the tailings-storage capacity for the Moab mill. The estimates indicated that seepage into the tailings pile during the early milling period of 1956−1974 was approximately 76 gpm. Because of revised ore-processing procedures, the seepage rate might have declined to approximately 48 gpm until closure of the mill.

After the mill closed, Canonie Environmental (Canonie 1994) used measured changes in water levels in the tailings pile from 1989 through 1994 to estimate the average seepage rate into the tailings pile. Canonie estimated the seepage at approximately 33 gpm during that period. Canonie also estimated that transient drainage of the pile would continue for approximately 56 years.

Using the finite-element code PORFLOW, ORNL (1998b) estimated that transient drainage from the tailings pile would require approximately 238 years to complete. ORNL also concluded that for a cover permeability of 1×10^{-7} centimeters per second (cm/s), the steady state drainage rate would be approximately 3.7 gpm, and that for a cover permeability of 1×10^{-8} cm/s the steady state drainage rate would be approximately 0.6 gpm.

Using results from the ORNL investigation combined with model-calibration results of their own, Armstrong et al. (1998) used a step function to describe drainage rate from the tailings pile. According to their work, the initial drainage rate consists of a flux of 40 gpm through the tailings pile during the period 1956−1996. For the post-milling period of 1997−2056 they used a flux of 6 gpm and 0.8 gpm to describe the seepage rates that would result from clay-cover hydraulic conductivities of 1×10^{-7}cm/s and 1×10^{-8}cm/s, respectively.

SRK (2000) conducted a drainage study of its own using the finite-element code SEEP/W (GEO SLOPE, International) to estimate the seepage flux versus time for the transient-drainage period. Their results showed that the initial drainage rate of 40 gpm would decay to a steady-state drainage rate of 11 gpm over a period exceeding 60 years.

Shepherd Miller, Inc. (SMI 2001) used a scalar adjustment of approximately ¼ to reduce the drainage flux presented in the SRK 2000 report. The SMI team reasoned that if the SRK-predicted drainage were correct, a physically unrealistic volume reduction would occur due to consolidation. On the basis of the scalar adjustment, SMI estimated that the initial drainage rate would be approximately 17 gpm and would decline to a steady-state drainage rate of about 4 gpm over approximately 24 years. The predicted steady-state drainage was estimated assuming a clay cover hydraulic conductivity of 1×10^{-7}cm/s.

3.9.1.4 Hydraulic Conductivity of the Tailings

Hydraulic conductivity values of the tailings were measured in the laboratory as part of the work performed by Dames & Moore between 1975 and 1981. The hydraulic conductivity results are summarized in Canonie 1994 and are reproduced in Table 3–10. The average hydraulic conductivity of the tailings in Table 3–10 is 9.25×10^{-5} cm/s. In addition, SRK (2000) indicates that according to their study, the unconsolidated sand tailings have a permeability of 5.57×10^{-5}cm/s, and the slimes have a permeability of 5.68×10^{-6}cm/s.

Table 3–10. Summary of Tailings Hydraulic Conductivity Results (after Canonie Environmental 1994; Table 2)

Borehole/ Sample ID	Depth (ft bgs)	Material Type	Hydraulic conductivity (cm/s)	Reference[b]
10	6.5	Fine Sand with Silt	3.87×10^{-6}	D&M (1975)
10	12.5	Fine Sand with Silt	1.93×10^{-6}	D&M (1975)
11	18.5	Fine Sand with Silt	1.93×10^{-5}	D&M (1975)
12	8.5	Fine Sand with Silt	3.38×10^{-5}	D&M (1975)
13	17.5	Fine Sand with Silt	1.35×10^{-5}	D&M (1975)
A-1	50	Silt with Fine Sand	1.26×10^{-6}	D&M (1981)
A-4	10	Fine to Med. Sand with Silt	5.31×10^{-4}	D&M (1981)
A-4	20	Fine Sand with Silt	5.22×10^{-5}	D&M (1981)
A-4	40	Fine Sand with Silt	7.05×10^{-5}	D&M (1981)
A-5	10	Fine to Med. Sand with Silt	1.55×10^{-5}	D&M (1981)
A-5	30	Fine to Med. Sand with Silt	2.03×10^{-4}	D&M (1981)
A-5	70	Silt with Clay	1.93×10^{-7}	D&M (1981)
A-6	30	Fine Sand with Silt	5.60×10^{-5}	D&M (1981)
A-6	40	Silt with Clay and Sand	2.32×10^{-6}	D&M (1981)
A-11	30	Silt with Clay	9.66×10^{-8}	D&M (1981)
A-11	40	Silt with Clay	9.66×10^{-8}	D&M (1981)
B-11	26.5	Fine Sand with Silt	4.36×10^{-4}	D&M (1979)
B-12	11.5	Fine to Med. Sand with Silt	4.98×10^{-4}	D&M (1979)
B-16	11.5	Fine to Med. Sand with Silt	6.07×10^{-4}	D&M (1979)
B-28	41.5	Silt with Clay and Sand	4.06×10^{-4}	D&M (1979)
Average Tailings Value			9.25×10^{-5}	

[a]bgs = below ground surface
[b]D&M = Dames & Moore

3.9.1.5 Subpile Soil

Subpile soils are potential long-term sources of ground water contamination because of chemical constituents that might have accumulated within them as pore fluids migrated through them. ORNL, SRK, and SMI sampled and characterized the subpile soils. The extraction method used by both SRK and SMI is an almost complete digestion using a strong acid and heat; these results probably overestimate the amount of contaminants that are available for leaching and are therefore conservative. There was no discussion of extraction method included in the ORNL report. Despite possible differences in extraction technique, results of different studies can be compared in a relative manner and are useful for identifying overall patterns in the data. Results of the subpile sampling performed by ORNL, SRK, and SMI are presented in Table 3–11.

3.9.2 Ground Water System

A construction summary of the wells and piezometers previously installed at the site is presented in Table 3–12. Locations are shown in Plate 1. The first wells drilled in the area were for oil exploration by Embar Oil in 1926 and 1928 with a total depth of approximately 300 and 5,345 ft, respectively. Records show a water well (C) was drilled in 1940 by the Grazing Service, in the vicinity of the area now covered at the north end of the tailings, presumably to provide water for livestock. Wells B and A were installed in 1954 and 1961, respectively, presumably to provide water to the milling operations. All of these wells were abandoned at some later date.

Table 3–11. Summary of Previous Subpile Soil Analyses

Sample ID[b] (includes depth)	Lithology	NH_4 (mg/kg)	As (mg/kg)	Ba (mg/kg)	Cu (mg/kg)	Fe (mg/kg)	Mn (mg/kg)	Mo (mg/kg)	Se (mg/kg)	SO_4 (mg/kg)	U (mg/kg)	V (mg/kg)
Oak Ridge National Laboratory Subpile Results												
PB-1-94	Alluvium	NA	3.1	75.8	15.4	NA	NA	20.5	0.62	NA	52.3	1720
PB-1-101	Alluvium	NA	5.5	187	10.7	NA	NA	0.98B	<0.2	NA	5.4	31.2
PB-2-90	Alluvium	NA	2.5	92.4	8.6	NA	NA	1.3B	0.3B	NA	3.6	33.2
PB-2-100	Alluvium	NA	5.5	195	10.3	NA	NA	1.3B	0.3B	NA	8.3	22.3
SRK Subpile Results (partial constituent list)												
AR2-44-45	Clayey gravel	52.1	1.52	104	18.2	5,700	241	0.14	0.33	NA	6.46	11.9
AR3-67-70	Red-brn sand	161	1.57	141	7.74	9,500	670	0.43	0.24	NA	4.21	4.75
AR4-46-47	Red sand w/gravel	2,467	15.6	63.7	33.5	9,980	205	38.7	0.95	NA	15.1	708
AR-4-46-50	Red sand w/gravel	NA	1.81	85.5	8.36	9,310	494	1.19	0.52	NA	4.94	14.4
Shepherd Miller Subpile Results												
AR-10-86-87	Silty sand	133	2.9	NA	NA	4,480	1,070	11.5	<0.2	18,810	56.5	1,550
AR4-67-70	Silty sand	608	0.8	NA	NA	5,420	272.2	<0.2	<0.2	4,710	2	11.4
AR4-85-90	Silty sand	3.6	4	NA	NA	13,900	377	0.3	0.3	1,410	0.9	21.5
AR7-100	Silty sand	1.1	2.8	NA	NA	8,880	218	1.6	0.4	1,296	4.6	14.9
AR7-70-75	Silty sand	1.2	1	NA	NA	5,940	287	0.4	<0.2	855	4.2	19.3
AR9-85-90	Silty sand	181	1.9	NA	NA	8,690	926	1.8	0.6	2,922	1.7	19
AR1-29-30	Gravelly sand	4.7	1.1	NA	NA	8,790	541	<0.2	<0.2	1,809	59.4	28.5
Shepherd Miller Background Soil Analyses												
SMI-BH01-11-13.5	Gravelly sand	2.3	0.9	89.8	3.1	4,670	301	<0.2	0.4	831	0.6	12.7
SMI-BH01-41-46	Gravelly sand	3.6	0.7	84.1	2.9	4,430	303	0.4	0.4	768	0.4	7.8
SRK Average Tailings Slimes Results												
Average of 7	Slimes	715	42	816	226	8,892	395	4.45	6.1	NA	135	953

[a] Extraction methods not provided for Oak Ridge data; SRK and SMI extraction method is almost complete digestion (EPA 3050); analytical methods EPA 6010B and 6020.

[b] Last digits in sample identification indicate depth (ft) below ground surface.

Notes:

NA = not ananlyzed; B = estimated value; NH_4 = ammonium; As = arsenic; Ba = barium; Cu = copper; Fe = iron; Mn = manganese; Mo = molybdenum; Se = selenium; SO_4 = sulfate; U = uranium; V = vanadium; mg/kg = milligrams per kilogram

Ammonium concentrations are total ammonium as NH_4

Slimes samples include locations AR4S-20-21, AR4S-21-25, AR4-30-35, AR8-21-22, AR8-22-25, AR8-25-35, AR8-40-45 (see Plate 1).

Table 3–12. Summary of Construction Information for Historical Wells

Location ID	Northing State Plane (ft)	Easting State Plane (ft)	Ground Elevation (ft)	Borehole Diameter (in.)	Top of Casing Elevation (ft)	Casing Diameter (in.)	Casing Length (ft)	Borehole Depth (ft)	Screen Depth (ft)	Screen Length (ft)	Zone of Completion	Status	Installed By	Installation Date	Location Comments
1	6664447.57	2185520.05	4030.6		4030.6		47	81.5				Decommissioned	D&M	28-Nov-72	
2	6664392.75	2185611.39	3986.2		3986.2		37	39				Decommissioned	D&M	29-Nov-72	
3	6664524.87	2185323.11	4025.7		4025.7		20	71.5				Decommissioned	D&M	30-Nov-72	
4	6665249.8	2185392.74	4030.8		4030.8		32.3	66.5				Decommissioned	D&M	01-Dec-72	
5	6665284.47	2185461.93	3992.2		3992.2		40	41.5				Decommissioned	D&M	05-Dec-72	
6	6666055.47	2184451.06	4031.5		4031.5		53	56.5				Decommissioned	D&M	04-Dec-72	
7	6664397.55	2183645.58	4029.2		4029.2		30	31.5				Decommissioned	D&M	05-Dec-72	
8	6664484.84	2185947.3	3967.3					62				Decommissioned	D&M	25-Apr-74	Converted to well MW-3
9	6665237.54	2186631.47	3962.5		3962.5		23	25				Decommissioned	D&M	24-Apr-74	
10	6663432.17	2184602.1	4004.2		4004.2		41	45				Decommissioned	D&M	16-Apr-74	
12	6663780.11	2184177.6	4030.1		4030.1		50	50.5				Decommissioned	D&M	18-Apr-74	
13	6665954.03	2185058.71	4032.4		4032.4		79	79.5				Decommissioned	D&M	18-Apr-74	
14	6666606.33	2183229.4	4016.3		4016.3		68	70				Decommissioned	D&M	22-Apr-74	
15	6666260.9	2183965.95	3997.1		3997.1		48	50				Decommissioned	D&M	22-Apr-74	
A	6664937.78	2183189.44	4045.9		4045.9		130	131	118	12		Decommissioned	M K & Co	14-Oct-61	Perforated casing; Yield 18 gpm
A-1	6666010.81	2184330.7	4044		4045.46	2	66.46	66	45	20	TA	Existing	D&M	07-Mar-77	Located on north side of tailings pile.
A-2 (3)	6666044.31	2184332.19	4031.2		4030.4	2	50.7	51.5	31.5	20		Decommissioned	D&M	07-Mar-77	
A-3 (2)	6666083.57	2184333.64	4020.5		4019.9	2	41.4	43	22	20		Decommissioned	D&M	07-Mar-77	
A-4	6665576.27	2185149.41	4040		4040.6	2	70.1	70.5	49.5	20	TA	Decommissioned	D&M	08-Mar-77	
A-5	6664428.09	2185416.88	4039.8		4040.7	2	81.9	82.5	61	20	TA	Decommissioned	D&M	08-Mar-77	
A-6	6663778.67	2184242.65	4040.7		4040.7	2	61	61	41	20		Decommissioned	D&M	08-Mar-77	
A-7	6665621.04	2185239.07	4002		4001.6	2	28.6	30.5	9	20	TA	Decommissioned	D&M	09-Mar-77	
A-8	6665648	2185261.93	3992.4		3992.2	2	19.8	20.5	7.5	12.5	TA	Decommissioned	D&M	09-Mar-77	
A-9	6664368.23	2185473.83	4007.9		4008	2	49.1	50	29	20	TA	Decommissioned	D&M	09-Mar-77	
A-10	6664327.74	2185537.13	3986.6		3986.7	2	19.6	20.5	7.5	12	TA	Decommissioned	D&M	09-Mar-77	
A-11	6663821.76	2184162.59	4028.8		4028.8	2	49.5	50.5	29.5	20		Decommissioned	D&M	09-Mar-77	
A-12	6665113.53	2183093.2	4045.7		4045.7	2	20	20.5	7.5	12.5		Decommissioned	D&M	09-Mar-77	
A-13	6665108.33	2183168.34				2		20.5	7.5	12.5		Decommissioned	D&M	10-Mar-77	
AEC	6666322.66	2182766.41										Existing	AEC		NW corner of DOE property.
AMM-1	6667297.85	2187877.59	3970	8	3972.02	4	59.32	65	17.25	39.75	AL	Existing	Atlas	14-Sep-88	NE corner of DOE property.
AMM-2	6664125.27	2186026.68	3965.5	8	3967.74	4	52.24	62	10	39.75	AL	Existing	Atlas	20-Sep-88	East of pile along road.
AMM-3	6663155.89	2185004.95	3965.7	8	3967.69	4	51.99	50	30	19.75	AL	Existing	Atlas	23-Sep-88	Near SE corner of pile.
AR1S	6665258.94	2183448.96	4046.3	7.75	4048.5	2	29.7	35	7.5	20	TA	Decommissioned	SRK	18-Apr-00	
AR2S	6664639.27	2183954.93	4040.6	7.75	4042.29	2	45.69	66.5	24	20	TA	Decommissioned	SRK	19-Apr-00	
AR4D	6665173.66	2184293.2	4030.9	7.75	4031.45	2	87.55	90	77	9.5	AL	Decommissioned	SRK	21-Apr-00	
AR5S	6665581.56	2184476.55	4038.5	7.75	4040.5	2	58	60	36	20	TA	Decommissioned	SRK	22-Apr-00	
AR7D	6665638.42	2183913.72	4046.6	7.75	4048.72	2	106.12	105	94	10	AL	Decommissioned	SRK	25-Apr-00	
AR7S	6665638.42	2183913.72	4046.6	7.75	4048.45	2	52.85	52	31	20	TA	Decommissioned	SRK	26-Apr-00	
AR8S	6664863.82	2184546.18	4031.3	7.75	4031.85	2	47.05	54	28	19	TA	Decommissioned	SRK	27-Apr-00	
AR9S	6663864.67	2184587.38	4049.6	7.75	4051.95	2	86.35	90	64	20	TA	Decommissioned	SRK	02-May-00	
AR10D	6664525.55	2185080.57	4046.3	7.75	4046.8	2	104.5	105	94	10	AL	Decommissioned	SRK	03-May-00	
AR11S	6664525.55	2185080.57	4046.3	7.75	4048.02	2	69.72	69	48	20	TA	Decommissioned	SRK	04-May-00	
AR12S	6664525.55	2185080.57	4046.3	7.75	4047.8	2	50.5	50	29	20	TA	Decommissioned	SRK	04-May-00	
AR13S	6665271.72	2184729.52	4032.1	7.75	4032.75	2	44.65	45	24	20	TA	Decommissioned	SRK	04-May-00	
ARCHES1978	6670663.98	2180116.16	4079.9	12		8	172	172	112	60	BR	Existing	NPS		Arches N. P. well; Perforated casing.

Table 3–12. Summary of Construction Information for Historical Wells (continued)

Location ID	Northing State Plane (ft)	Easting State Plane (ft)	Ground Elevation (ft)	Borehole Diameter (in.)	Top of Casing Elevation (ft)	Casing Diameter (in.)	Casing Length (ft)	Borehole Depth (ft)	Screen Depth (ft)	Screen Length (ft)	Zone of Completion	Status	Installed By	Installation Date	Location Comments
ATP-1-D	6664508.58	2185981.58	3968.2	8	3970.73	2	402.53	406	395	5	AL	Existing	D&M	30-Nov-81	
ATP-1-ID	6664508.58	2185981.58	3968.2	8	3970.87	2	302.67	406	295	5	AL	Existing	D&M	30-Nov-81	
ATP-1-IS	6664508.58	2185981.58	3968.2	8	3971	2	222.8	406	215	5	AL	Existing	D&M	30-Nov-81	
ATP-1-S	6664508.58	2185981.58	3968.2	8	3971.14	2	157.94	406	145	10	AL	Existing	D&M	30-Nov-81	
ATP-2-D	6663829.7	2185459.8	3964.4		3967.05	2	92.65	97	80	15	AL	Existing	D&M	01-Jun-73	
ATP-2-S	6663829.47	2185459.58	3964.4		3967.04	2	40.64	97	29	10	AL	Existing	D&M	01-Jun-73	
ATP-3	6666652.75	2183965.46	3996.9		3998.49	4	61.59	65	53	10	AL	Existing	D&M	12-Jun-82	
B	6666313.49	2182907.32	4037.9		4037.9	5.37	114	114	104	10		Decommissioned	AEC	10-May-54	Perforated casing; Yield 11 gpm
B-1 (15)	6664789.32	2185537.85	4039		4040.8	2	61.8	115.5	50	10		Decommissioned	D&M	15-Jun-78	
B-2 (14)	6664005.02	2185124.63	4039		4040.3	2	41.3	120	30	10		Decommissioned	D&M	16-Jun-78	Well depth 114.9 ft. on log
B-3	6664020.18	2185135.17	4039					110				Decommissioned	D&M	18-Jun-78	No standpipe/piezometer installed
B-4 (17)	6665981	2183793.93	4043.8		4044.63	2	80.83	110	70	10		Decommissioned	D&M	18-Jun-76	
B-5 (16)	6665984.45	2184871.91	4040		4040.5	2	80.5	80.5	70	10		Decommissioned	D&M	19-Jun-78	
B-6	6666002.04	2184850.48	4040					80.5				Decommissioned	D&M	19-Jun-78	No standpipe/piezometer installed
B-7	6663632.34	2184627	4046.3		4057.3	0.75	93	119	80	2	TA	Decommissioned	D&M	31-Oct-78	1.5-inch "hydro-tip" piezometer
B-8	6663556.68	2184625.72	4040.4		4039.4	0.75	67	79.5	66	2	TA	Decommissioned	D&M	20-Nov-78	1.5-inch "hydro-tip" piezometer
B-9	6663371.38	2184649.08	3987.7		3987.5	0.75	14.8	40.5	13	2	TA	Decommissioned	D&M	21-Nov-78	1.5-inch "hydro-tip" piezometer
B-10	6664065.59	2185076.86	4045.1		4057.6		93.5	115.5	79	2	TA	Decommissioned	D&M	06-Nov-78	1.5-inch "hydro-tip" piezometer
B-11	6663976.8	2185211.64	4010.6		4010.9	0.75	50.6	58	48.3	2	TA	Decommissioned	D&M	15-Nov-78	1.5-inch "hydro-tip" piezometer
B-12	6663929.51	2185279.1	3985.7		3986.5	0.75	26.8	45.5	24	2	AL	Decommissioned	D&M	16-Nov-78	1.5-inch "hydro-tip" piezometer
B-13	6664756.25	2185473.11	4046.2		4054.9	0.75	91.8	100.5	81.1	2	TA	Decommissioned	D&M	09-Nov-78	1.5-inch "hydro-tip" piezometer
B-14	6664857.06	2185655.15	3993.1		3993.1	0.75	29.8	70	27.8	2	AL	Decommissioned	D&M	17-Nov-78	1.5-inch "hydro-tip" piezometer
B-15	6665915.44	2184813.95	4046.4		4055.2	0.75	69.8	120.5	59	2	TA	Decommissioned	D&M	14-Nov-78	1.5-inch "hydro-tip" piezometer
B-16	6665979.28	2184870.29	4043.8		4044.01	0.75	22.71	30.5	20.5	2	TA	Existing	D&M	18-Nov-78	1.5-inch "hydro-tip" piezometer
B-17	6666139.22	2184997.24	3987.1		3987.1	0.75	8.5	66.7	6.5	2	AL	Decommissioned	D&M	22-Nov-78	1.5-inch "hydro-tip" piezometer
B-18	6663654.86	2184793.82	4046					115.5				Decommissioned	D&M	20-Oct-78	No standpipe/piezometer installed
B-19	6663837.93	2184919.47	4046					115.5				Decommissioned	D&M	23-Oct-78	No standpipe/piezometer installed
B-20	6664273.17	2185214.13	4046					112				Decommissioned	D&M	25-Oct-78	No standpipe/piezometer installed
B-21	6665008.56	2185344.51	4046					102.5				Decommissioned	D&M	26-Oct-78	No standpipe/piezometer installed
B-22	6665230.86	2185229.14	4046					122.5				Decommissioned	D&M	20-Oct-78	No standpipe/piezometer installed
B-23	6665789.3	2184947.63	4046					100.5				Decommissioned	D&M	28-Oct-78	No standpipe/piezometer installed
B-24	6665936.16	2184522.95	4046					90.5				Decommissioned	D&M	29-Oct-78	No standpipe/piezometer installed
B-25	6665752.96	2183475.97	4046					55.5				Decommissioned	D&M	29-Oct-78	No standpipe/piezometer installed
B-26	6664765.98	2183366.64	4046					130				Decommissioned	D&M	18-Oct-78	No standpipe/piezometer installed
B-27	6664104.7	2183947.72	4046					85.5				Decommissioned	D&M	19-Oct-78	No standpipe/piezometer installed
B-28	6665928.45	2184349.21	4056.7		4056.92	0.75	55.22	112	53	2	TA	Existing	D&M	28-Nov-78	1.5-inch "hydro-tip" piezometer
BAA-2	6664468.5	2185362.79	4055.3		4058.7	0.75	65	97	59.6	2	TA	Decommissioned	D&M	19-Jul-84	1.5-inch "hydro-tip" piezometer; see boring AA1-84
BAA-3	6664496.46	2185327.4	4053.4		4055.8	0.75	63.5	91.5	59.1	2	TA	Decommissioned	D&M	13-Jul-84	1.5-inch "hydro-tip" piezometer; see boring AA2-84
BAA-4	6664544.43	2185258.63	4052.3		4055.1	0.75	56.5	54.5	51.7	2	TA	Decommissioned	D&M	10-Jul-84	1.5-inch "hydro-tip" piezometer; see boring AA3-84
C	6666060	2183525	3980		3980	8	67	67	53	14	AL	Decommissioned	Grazing Service, DOI	17-Jan-40	Perforated casing; yield 20 gpm; buried under tailings disposal area.
C-1						6		117			BR	Decommissioned	NPS	29-Feb-40	Not perforated; Open hole?
C-2			4090		4090	8	123	123	96	27	BR	Decommissioned	NPS	05-Jan-59	Arches N. P. well; Perforated casing.

Table 3–12. Summary of Construction Information for Historical Wells (continued)

Location ID	Northing State Plane (ft)	Easting State Plane (ft)	Ground Elevation (ft)	Borehole Diameter (in.)	Top of Casing Elevation (ft)	Casing Diameter (in.)	Casing Length (ft)	Borehole Depth (ft)	Screen Depth (ft)	Screen Length (ft)	Zone of Completion	Status	Installed By	Installation Date	Location Comments
D-1	6664614.38	2183456.61	4058		4057.4	0.75	37.4	50	36.5	1.5	NR	Decommissioned	D&M		No boring log
D-2	6664227.21	2186294.77	3954.167		4057.7	0.75	137.533	50	32.5	1.5	NR	Decommissioned	D&M		No boring log
D-3	6663841.27	2184137.92	4058		4057.8	0.75	33.8	50	32.5	1.5	NR	Decommissioned	D&M		No boring log
EE-2	6665946.18	2184355.51	4056.9		4057.19	0.75	62.89	76.5	60.6	2	NR	Existing	D&M	27-Jul-84	1.5-inch "hydro-tip" piezometer; see boring EE1-84 top of pile.
EE-3	6665906.98	2184354.2	4056.5		4058.82	0.75	64.52	71.5	60.2	2	NR	Existing	D&M	25-Jul-84	1.5-inch "hydro-tip" piezometer; see boring EE2-84 top of pile.
EE-4	6665835.11	2184351.71	4053.6		4054	0.75	41.6	41.5	39.2	2	TA	Decommissioned	D&M	20-Jul-84	1.5-inch "hydro-tip" piezometer; see boring EE3-84
Embar Oil Big Six	6660811	2185728	3965			12		5345				Decommissioned	Embar Oil	02-Mar-28	Abandoned oil well.
Embar Oil No. 1	6667142	2188877						300				Decommissioned	Embar Oil	01-Jan-26	Abandoned oil well. Operator and well number: Embar Oil - Big Six Oil Companies, No. 1
M1	6658105.42	2191088.86	3965.09		3966.1						AL	Existing	MWP		Matheson Wetlands Preserve
M4	6659625.58	2192123.95	3965.3		3966.6						AL	Existing	MWP		Matheson Wetlands Preserve
M5	6659985.82	2192144.58	3964.56		3965.45						AL	Existing	MWP		Matheson Wetlands Preserve
M6	6660053.1	2191662.7	3961.67		3962.63						AL	Existing	MWP		Matheson Wetlands Preserve
M8	6665642.82	2188420.66	3957.75		3963.41						AL	Existing	MWP		Matheson Wetlands Preserve
M9	6664462	2187523									AL	Existing	MWP		Matheson Wetlands Preserve
M10	6658900	2188471									AL	Existing	MWP		Matheson Wetlands Preserve
M11-4.8	6660140.15	2187730.64	3963.27		3964.61						AL	Existing	MWP		Matheson Wetlands Preserve
M11-6	6660134.58	2187742.24	3963.71		3963.75						AL	Existing	MWP		Matheson Wetlands Preserve
M11-7.0	6660139.56	2187729.37	3963.19		3964.56						AL	Existing	MWP		Matheson Wetlands Preserve
M11-12	6660133.24	2187734.21	3963.46		3964.16							Existing	MWP		Matheson Wetlands Preserve
M11-14.0	6660139.08	2187728.12	3963.17		3964.57						AL	Existing	MWP		Matheson Wetlands Preserve
M12											AL	Existing	MWP		Matheson Wetlands Preserve
M13	6663160	2190492									AL	Existing	MWP		Matheson Wetlands Preserve
M16	6656288	2191700.91	3961.63		3962.65						AL	Existing	MWP		Matheson Wetlands Preserve
MW 1-S	6663218.65	2185112.13	3962.616		3964.42							Decommissioned			
MW-1	6665679.97	2186810.41	3966.218		3968.429						AL	Decommissioned			
MW-1-R	6663218.21	2185112.08	3962.6		3964.35	4	14.75	15	3	10	AL	Existing	D & M	13-Jun-82	
MW-2-R	6664209.52	2185787.11	3965.2		3966.7	2	11.5	11	2	8	AL	Existing	D & M	13-Jun-82	
MW-3	6664526.68	2185907.13	3968.1		3969.21	4	56.11	62			AL	Existing	D & M	25-Apr-74	See borehole 8
N2-1.5	6661863.51	2189784.94	3962.14		3962.54						AL	Existing	MWP		Matheson Wetlands Preserve
N2-12.8	6661862.65	2189780.9	3962		3963.11						AL	Existing	MWP		Matheson Wetlands Preserve
N2-4.3	6661861.32	2189783.42	3962.14		3962.87						AL	Existing	MWP		Matheson Wetlands Preserve
N2-6.5	6661861.95	2189782.23	3961.97		3963.01						AL	Existing	MWP		Matheson Wetlands Preserve
N3-4.3	6663417.15	2191403.56	3964.17		3964.71						AL	Existing	MWP		Matheson Wetlands Preserve
N3-8.3	6663417.62	2191404.35	3964.09		3965.03						AL	Existing	MWP		Matheson Wetlands Preserve
N4-12.0	6658292.45	2191112.66	3961.44		3963.27						AL	Existing	MWP		Matheson Wetlands Preserve
N4-3.2	6658293.87	2191110.63	3961.39		3962.35						AL	Existing	MWP		Matheson Wetlands Preserve
N4-6.5	6658292.51	2191111.16	3961.37		3962.66						AL	Existing	MWP		Matheson Wetlands Preserve
N5-10	6660289.71	2192132.63	3964.42		3965.51						AL	Existing	MWP		Matheson Wetlands Preserve
N5-14	6660288.5	2192130.47	3964.43		3965.59						AL	Existing	MWP		Matheson Wetlands Preserve
N5-4.3OLD	6660287.6	2192133.35	3964.41		3965.53						AL	Existing	MWP		Matheson Wetlands Preserve
N5-4.4NEW	6660289.09	2192131.65	3964.44		3965.43						AL	Existing	MWP		Matheson Wetlands Preserve
N5-7.2	6660286.84	2192131.14	3964.56		3965.82						AL	Existing	MWP		Matheson Wetlands Preserve

Table 3–12. Summary of Construction Information for Historical Wells (continued)

Location ID	Northing State Plane (ft)	Easting State Plane (ft)	Ground Elevation (ft)	Borehole Diameter (in.)	Top of Casing Elevation (ft)	Casing Diameter (in.)	Casing Length (ft)	Borehole Depth (ft)	Screen Depth (ft)	Screen Length (ft)	Zone of Completion	Status	Installed By	Installation Date	Location Comments
N6-4.5	6666026.38	2189886.83	3960.85		3962.74						AL	Existing	MWP		Matheson Wetlands Preserve
N6-6.4	6666027.04	2189887.36	3960.72		3962.69						AL	Existing	MWP		Matheson Wetlands Preserve
N7-10	6655960.93	2190421.01	3962.84		3964.41						AL	Existing	MWP		Matheson Wetlands Preserve
N7-11	6655959.38	2190418.89	3963.1		3963.84						AL	Existing	MWP		Matheson Wetlands Preserve
N7-4	6655961.48	2190419.83	3962.77		3964.32						AL	Existing	MWP		Matheson Wetlands Preserve
N7-7	6655960.11	2190420.28	3963.08		3964.37						AL	Existing	MWP		Matheson Wetlands Preserve
N8-10	6660114.59	2188737.83	3963.4		3964.94						AL	Existing	MWP		Matheson Wetlands Preserve
N8-14	6660112.71	2188736.53	3963.48		3964.91						AL	Existing	MWP		Matheson Wetlands Preserve
N8-3	6660114.09	2188738.96	3963.44		3965.03						AL	Existing	MWP		Matheson Wetlands Preserve
N8-6	6660113.58	2188737.24	3963.46		3964.79						AL	Existing	MWP		Matheson Wetlands Preserve
NE-MILL	6666736.6	2186780.45	3979		3981.45						NR	Existing	Atlas		
OLD BYRD	6660728.94	2191798.01	3961		3964.12						AL	Existing	MWP		Matheson Wetlands Preserve
OW-1	6663838.85	2185471.23	3964.4	2	3966.94	1	32.54	30	20	10	AL	Existing	ORNL	21-Nov-97	Observation well
OW-2	6663845.27	2185479.32	3964.5	2	3966.85	1	32.35	30	20	10	AL	Existing	ORNL	21-Nov-97	Observation well
OW-3	6663827.07	2185444.7	3964.7	4	3966.2	1	39.5	38	28	10	AL	Existing	ORNL	02-Dec-97	Observation well
OW-4	6663828.7	2185434.41	3964.4	4	3965.5	1	39.1	38	28	10	AL	Existing	ORNL	02-Dec-97	Observation well
P1	6665993.96	2184347.5				0.75						Decommissioned	D & M		1.5-in open-tube "casagrande" piezometer; P1(HPP-1?)
P2	6665547.13	2185164.81				0.75						Decommissioned	D & M		1.5-in open-tube "casagrande" piezometer; P2(HPP-2?)
P3	6664440.77	2185424.37				0.75						Decommissioned	D & M		1.5-in open-tube "casagrande" piezometer; P3(HPP-3?)
PW-1	6664574.18	2185333.63	4058.3		4058.13	5	97.83	120	49.37	40.07	TA	Existing	WTI	16-Mar-90	
PW-2	6664416.07	2185033.04	4050.9		4050.9	4	82	120	35.8	40	TA	Decommissioned	WTI	30-Jun-90	Data from table; No logs.
PW-3	6664115.9	2184523.4	4052.1		4056.26	4	72.66	75	8.5	60	TA	Existing	WTI	15-Jun-90	
PW-4	6665078.05	2185077.64	4054		4055.82	4	70.32	71.5	8.2	60.3	TA	Existing	WTI	21-Jun-90	
PW-4-OB-A	6665043.51	2185063.3	4052.8		4055.28	4	41.08	39	8.3	30.3	TA	Existing	WTI	16-Jun-90	
PW-4-OB-B	6665069.51	2185063.49	4053.4		4054.19	4	69.59	70	8.5	60.3	TA	Existing	WTI	20-Jun-90	
PW-5	6664203.44	2184836.04	4055.4		4056.62	4	75.82	75	4.3	70.3	TA	Existing	WTI	17-Jun-90	
PW-6	6664679.13	2185174.56	4052.9		4053.01	4	73.71	75	3.6	70	TA	Existing	WTI	18-Jun-90	
PW-7	6664860.86	2185347.83	4056.5		4058.42	4	80.02	80.5	7.8	70.3	TA	Existing	WTI	19-Jun-90	
PW-8	6665212.1	2184985.3	4051.1		4055.09	4	72.99	71.5	8.8	60.2	TA	Existing	WTI	22-Jun-90	
PW-9	6664289.78	2184934.59	4054.4		4054.74	4	69.34	71.5	8.6	60.4	TA	Existing	WTI	23-Jun-90	
PW-10	6664108.45	2184770.78	4055.1		4056.66	4	71.86	73.5	10	60.3	TA	Existing	WTI	24-Jun-90	
PW-11	6663982.75	2184654.15	4054		4058.12	4	69.12	66.5	5	60	TA	Existing	WTI	25-Jun-90	
PW-12	6664877.69	2185132.51	4051.8		4053.25	4	70.35	71.5	8.6	60.3	TA	Existing	WTI	27-Jun-90	
PW-13	6665907.2	2184134.35	4056		4059.08	4	49.88	51	6.5	40.3	TA	Existing	WTI	28-Jun-90	
RW-01	6667744.37	2182253.43	4022.3	9	4021.81	2	78.51	81	69	10	AL	Existing	ORNL	09-Dec-97	National Park Service easement on BLM land.
SMI-MW01	6665679.85	2186810.55	3966.5	8.25	3968.32	2	31.51	30.5	14.41	15	AL	Existing	SMI	02-Oct-00	
SMI-PW01	6664474.71	2186192.88	3966.4	10	3968.45	4	62.26	63	20.09	40	AL	Existing	SMI	26-Oct-00	
SMI-PW02	6663621.54	2185811.47	3965.6	10	3967.48	4	62.18	63	20.04	40	AL	Existing	SMI	01-Nov-00	
SMI-PW03	6666133.74	2186218.94	3973.1	10	3975.04	4	62.43	63.8	20.23	40	AL	Existing	SMI	13-Oct-00	
SMI-PZ1D2	6664475.11	2186170.77	3966.4	10	3968.26	2	76.89	83	69.75	5	AL	Existing	SMI	25-Oct-00	
SMI-PZ1M	6664493.6	2186178.71	3966.3	10	3968.29	2	62.79	64	55.53	5	AL	Existing	SMI	28-Oct-00	
SMI-PZ1S	6664492.79	2186200.77	3966.7	10	3969.13	2	21.57	20	13.86	5	AL	Existing	SMI	26-Sep-00	
SMI-PZ2D	6663620.52	2185789.29	3965.2	10	3967.38	2	80.67	82	73.19	5	AL	Existing	SMI	29-Oct-00	
SMI-PZ2M1	6663640.06	2185820.1	3965.5	10	3967.5	2	62.27	64	55	5	AL	Existing	SMI	02-Nov-00	

Table 3–12. Summary of Construction Information for Historical Wells (continued)

Location ID	Northing State Plane (ft)	Easting State Plane (ft)	Ground Elevation (ft)	Borehole Diameter (in.)	Top of Casing Elevation (ft)	Casing Diameter (in.)	Casing Length (ft)	Borehole Depth (ft)	Screen Depth (ft)	Screen Length (ft)	Zone of Completion	Status	Installed By	Installation Date	Location Comments
SMI-PZ2M2	6663639.31	2185796.6	3965.1	10	3967.18	2	62.44	64	55.08	5	AL	Existing	SMI	31-Oct-00	
SMI-PZ3D2	6666136.01	2186245.73	3973.3	10	3975.13	2	82.38	83	75.28	5	AL	Existing	SMI	17-Oct-00	
SMI-PZ3M	6666120.53	2186233.11	3973.3	10	3975.23	2	62.63	63	54.8	5	AL	Existing	SMI	15-Oct-00	
SMI-PZ3S	6666105.05	2186220.2	3972.9	10	3975.03	2	29.36	28	21.94	5	AL	Existing	SMI	03-Oct-00	
SS-1	6662225.03	2189492.97	3963.57		3964.55						AL	Existing	MWP		Matheson Wetlands Preserve
SS-2	6662593.18	2189340.65	3962.9		3963.99						AL	Existing	MWP		Matheson Wetlands Preserve
SS-3	6661784.93	2189652.59	3960.9		3962.84						AL	Existing	MWP		Matheson Wetlands Preserve
SS-4	6662038.19	2189595.88	3961.87		3962.88						AL	Existing	MWP		Matheson Wetlands Preserve
SS-5	6663322.15	2189128.38	3963.09		3967.29						AL	Existing	MWP		Matheson Wetlands Preserve
SS-6	6665054.12	2188481.96	3963.36		3964.33						AL	Existing	MWP		Matheson Wetlands Preserve
SS-7	6662917.05	2189294.41	3963.1		3964.07						AL	Existing	MWP		Matheson Wetlands Preserve
TH-20	6666165.65	2183757.37	3998		3998.6		56.6	60.5			AL	Decommissioned	WCC	31-Mar-81	Open well piezometer
TH-21	6666225.68	2184690.45	3985		3988		62	60			AL	Decommissioned	WCC	31-Mar-81	Open well piezometer
TH-22	6666240.66	2185197.17	3987		3979.1		31.6	39.5			AL	Decommissioned	WCC	01-Apr-81	Open well piezometer
TH-23	6665963.21	2182940.82	4011.363		4012.299		60.936	60			AL	Decommissioned	WCC	01-Apr-81	Open well piezometer
TH-24	6666526.54	2182901.4	4004.357		4005.238		60.881	60			AL	Decommissioned	WCC	02-Apr-81	Open well piezometer
TH-25	6666738.07	2184987.64	3989	7	3990.04	2	41.04	55			AL	Existing	WCC	31-Mar-81	Open well piezometer
TH-26	6666502.04	2184089.5	3989		3990.8		45.8	60.5			AL	Decommissioned	WCC	03-Apr-81	Open well piezometer
TP-01	6666339.91	2187357.59	3967.6	2	3969.39	1	25.79	24	19	5	AL	Existing	ORNL	17-Nov-97	
TP-02	6665915.09	2186979.65	3973.9	2	3975.55	1	33.65	32	27	5	AL	Existing	ORNL	18-Nov-97	
TP-03	6665411.87	2186637.42	3961.11		3960.928	1	23.818	24	19	5	AL	Decommissioned	ORNL	19-Nov-97	
TP-04	6664882.65	2186385.91	3969.94		3969.94	1	24	24	19	5	AL	Decommissioned	ORNL	19-Nov-97	
TP-05	6664860.15	2186065.39	3960.82		3960.82	1	16	16	11	5	AL	Decommissioned	ORNL	19-Nov-97	
TP-06	6662810.3	2185082.79	3962.2	2	3961.71	1	31.51	32	27	5	AL	Existing	ORNL	20-Nov-97	
TP-07	6662801.92	2185522.89	3964.6	2	3965.72	1	30.62	29.5	24.5	5	AL	Existing	ORNL	20-Nov-97	
TP-08	6663276.44	2185682.11	3966.3	2	3966.57	1	31.77	31.5	26.5	5	AL	Existing	ORNL	20-Nov-97	
TP-09	6663761.95	2185879.04	3965.8	2	3967.38	1	29.58	28	23	5	AL	Existing	ORNL	20-Nov-97	
TP-10	6664251.68	2186083.34	3964.04		3963.754	1	25.714	26	21	5	AL	Decommissioned	ORNL	20-Nov-97	
TP-11	6666817.18	2187966.5	3966.1	2	3967.51	1	33.41	32	27	5	AL	Existing	ORNL	21-Nov-97	
TP-12	6664641.63	2186344.66	3965.54		3965.54	1	20	20	15	5	AL	Decommissioned	ORNL	22-Nov-97	
TP-13	6664032.59	2186195.65	3965.88		3965.88	1	21	21	16	5	AL	Decommissioned	ORNL	22-Nov-97	
TP-14	6663558.06	2186071.89	3964.92		3964.92	1	21	21	16	5	AL	Decommissioned	ORNL	22-Nov-97	
TP-15	6663035.53	2185991.14	3963.94		3963.94	1	31	31	26	5	AL	Decommissioned	ORNL	22-Nov-97	
TP-16	6662561.5	2185933.88	3962.77		3962.77	1	27	27	22	5	AL	Decommissioned	ORNL	23-Nov-97	
TP-17	6661878.78	2185893.04	3964	2	3963.69	1	31.69	32	27	5	AL	Existing	ORNL	23-Nov-97	
TP-18	6661174.68	2186167.72	3963.9	2	3963.63	1	23.73	24	19	5	AL	Existing	ORNL	23-Nov-97	
TP-19	6660471.83	2186368.94	3962.3	2	3962.17	1	31.87	32	27	5	AL	Existing	ORNL	24-Nov-97	
TP-20	6662194.49	2185361.22	3966.5	2	3967.55	1	37.05	36	31	5	AL	Existing	ORNL	24-Nov-97	
TP-21	6659975.56	2186400.29	3963.7	2	3964.64	1	25.44	24.5	19.5	5	AL	Existing	ORNL	24-Nov-97	
W1-10	6663715.52	2187664.96	3964.36		3965.56						AL	Existing	MWP		Matheson Wetlands Preserve
W1-3.5	6663713.2	2187666.19	3964.32		3965.39						AL	Existing	MWP		Matheson Wetlands Preserve
W1-4.3	6663709.44	2187679.65	3964.18		3965.39						AL	Existing	MWP		Matheson Wetlands Preserve
W1-7	6663714.37	2187665.75	3964.32		3965.43						AL	Existing	MWP		Matheson Wetlands Preserve
W6	6658043.33	2188956.74	3961.82		3962.97						AL	Existing	MWP		Matheson Wetlands Preserve

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3.9.2.1 Monitor Wells

Ground water wells to monitor water quality have been installed on the site over a series of 10 different investigations. The first ground water monitor wells and piezometers associated with site characterization were installed after 1970. A summary of the significant investigations is presented below. All monitoring locations are shown on Plate 1.

A number of borings (labeled 1 through 15 on Plate 1), drilled by Dames & Moore from 1972 through 1974, were completed as piezometers. These piezometers represent the first characterization work completed on site. Subsequent investigations completed by Dames & Moore in 1976 included the installation of piezometers P-1 through P-3, and in 1977 piezometers A-1 through A-14 were installed. Piezometers B-1 through B-28 were installed during the 1978 Dames & Moore investigation.

In 1981 Woodward Clyde Consultants installed piezometers TH-20 through TH-28 along the northern boundary of the tailings pile. Dames & Moore installed the ATP series of monitor wells across the site in 1982, and in 1984 installed the AA, BAA, and EE series of wells. Atlas installed well AMM-1 at the northeast corner of the site and wells AMM-2 and AMM-3 along the Colorado River in 1988. In 1990 Western Technologies, Inc., installed monitor wells PW-1 through PW-13. Shepard Miller, Inc., installed the SMI series of wells during the 2000 field investigation.

3.9.2.2 Monitoring

Ground water monitoring from 1979 to 1988 involved intermittent sampling of 12 on-site wells and the off-site well at Arches National Park; two piezometers were also sampled (Canonie 1994). In 1984, monitoring of the Arches well was discontinued; two new wells (MW-1-R and MW-2-R) were installed to replace two existing wells (MW-1 and MW-2). Three new wells—AMM-1, AMM-2, and AMM-3—were installed in 1988. In 1989, monitoring of all wells except AMM-1, AMM-2, AMM-3, and ATP-2-S was discontinued. AMM-1 was established as the background well and AMM-2 and AMM-3 as point of compliance wells (Canonie 1994).

Between 1979 and 1988 monitoring samples were collected quarterly (Canonie 1994). Analytes included arsenic, barium, beryllium, cadmium, chloride, chromium, copper, cyanide, iron, lead, manganese, molybdenum, nickel, nitrate, potassium, selenium, silver, sodium, sulfate, vanadium, and zinc. Radionuclides included gross alpha, gross beta, lead-210, polonium-210, radium-226, radium-228, thorium-230, and elemental uranium. Other analyses included acetone, conductivity, methylene chloride, pH, and TDS. Not all wells were analyzed for all constituents for all sampling events. After 1988 and through at least 1994, numerous constituents were eliminated from the monitoring list. These included arsenic, iron, manganese, potassium, conductivity, thorium-230, lead-210, and polonium-210 (Canonie 1994). After 1994, it appears that most of the analytical work was done in conjunction with other studies that were focused on reclamation issues, as opposed to regular compliance monitoring.

3.9.2.3 Aquifer Parameters

In some instances, hydrologic testing of the alluvial aquifer (in the form of laboratory permeability testing, field permeability testing, slug tests, or aquifer tests) was completed as part of the previous site characterization investigations. A summary of the data source, testing location, test type, and results are presented in Table 3–13. Locations are presented in Plate 1.

Laboratory permeability tests were completed on samples collected from borings 8, 11, and 15. All samples were collected from the alluvial aquifer. Results indicated that the horizontal hydraulic conductivity ranged from 0.04 to 0.8 ft/day. Field permeability tests conducted at borings 8 and 15 indicated that conductivity ranged from 0.6 to 21.9 ft/day.

Another round of laboratory permeability tests was conducted on samples collected from the installation of piezometers A-1, A-3, and A-6. These samples were also collected from the alluvial aquifer, and the results indicated that conductivity ranged from 0.004 to 5.5 ft/day.

A number of slug tests were conducted using piezometers TH-21, TH-24, and TH-26. The results suggested that conductivity of the alluvial aquifer ranged from 1.9 to 7.2 ft/day, and the conductivity of the shallow unit ranged from 0.9 to 1.3 ft/day. An aquifer test was conducted using well ATP-2-S as the pumping well. The results indicated that hydraulic conductivity was 22 ft/day.

A number of aquifer and slug tests were completed at various locations during the Shepard Miller 2000 investigation. Analysis of the aquifer test drawdown data indicated that horizontal hydraulic conductivity of the alluvial aquifer ranges from 93.1 to 202 ft/day, and the vertical hydraulic conductivity ranged from 9.7 to 29.6 ft/day. Analysis of slug test data indicated that horizontal conductivity of the aquifer ranged from 2.1 to 80.2 ft/day. The results of slug test data from the aquifer indicates a horizontal conductivity of 19.4 to 49.4 ft/day.

3.9.3 Surface Water

During the early operational period of the Moab mill, monitoring requirements were less rigorous than in subsequent years. Monitoring at the plant largely focused on air monitoring of radionuclides. The U.S. Department of Health, Education, and Welfare conducted one of the first monitoring studies of the Colorado River in the vicinity of the Moab site (HEW 1961) as part of the Colorado River Basin Water Control Project. One of the project objectives was to identify the most pressing pollutant sources along the river and to secure necessary remedial actions to address them. Monitoring points were set up along the Colorado River upstream and downstream of the mill. During that time effluent from ore processing was being discharged to the river. Effluent discharge was estimated at 545 to 645 gpm (HEW 1961). Table 3–14 summarizes the chemistry of the effluent prior to discharge to the river (sample collected in the ditch between the pond and the river), an immediately upstream location (0.6 mile upstream from discharge) and the first sampling point downstream of the discharge point (2 miles downstream). Results were obtained for two sampling cycles.

Table 3–13. Summary of Historical Results of Hydrologic Tests of the Alluvial Aquifer

Source	Location	Piezometer Well/Boring	Location Description	Test Type	Depth (bgs)	K_h (ft/day)	K_v (ft/day)
D & M 1975	8	boring	Vicinity of current SMI-PW01	lab perm	5.5	0.04	
D & M 1975	8	boring	Vicinity of current SMI-PW01	lab perm	18.5	0.3	
D & M 1975	11	boring	Southern tip of pile	lab perm	69.5	0.8	
D & M 1975	15	boring	Vicinity of current ATP-3	lab perm	8.5	1	
D & M 1975	15	boring	Vicinity of current ATP-3	lab perm	13.5	0.3	
D & M 1975	15	boring	Vicinity of current ATP-3	lab perm	18.5	0.6	
D & M 1975	15	boring	Vicinity of current ATP-3	lab perm	39.5	0.5	
D & M 1975	8	boring	Vicinity of current SMI-PW01	field perm	13	2.3	
D & M 1975	8	boring	Vicinity of current SMI-PW01	field perm	18	3.8	
D & M 1975	8	boring	Vicinity of current SMI-PW01	field perm	28	21.9	
D & M 1975	15	boring	Vicinity of current ATP-3	field perm	8	0.7	
D & M 1975	15	boring	Vicinity of current ATP-3	field perm	13	1.2	
D & M 1975	15	boring	Vicinity of current ATP-3	field perm	18	0.6	
D & M 1981	A-1	piezometer	On top of north end of pile	lab perm	60.5	0.004	
D & M 1981	A-3	piezometer	Side of north end of pile	lab perm	43	0.2	
D & M 1981	A-6	piezometer	Side near southern end of pile	lab perm	62	5.5	
D & M 1982	TH-21	piezometer	Base of the north end of pile	slug test	24.4−60	7.2	
D & M 1982	TH-21	piezometer	Base of the north end of pile	slug test	24.8−60	6.7	
D & M 1982	TH-21	piezometer	Base of the north end of pile	slug test	24.8−60	3.6	
D & M 1982	TH-21	piezometer	Base of the north end of pile	slug test	24.9−60	2.2	
D & M 1982	TH-24	piezometer	Northeast corner of the site	slug test	38.5−60	1.2	
D & M 1982	TH-24	piezometer	Northeast corner of the site	slug test	38.8−60	1.3	
D & M 1982	TH-24	piezometer	Northeast corner of the site	slug test	38.8−60	0.9	
D & M 1982	TH-26	piezometer	Vicinity of current ATP-3	slug test	22.3−60	7.1	
D & M 1982	TH-26	piezometer	Vicinity of current ATP-3	slug test	23.2−60	3.9	
D & M 1982	TH-26	piezometer	Vicinity of current ATP-3	slug test	23.4−60	2.6	
D & M 1982	TH-26	piezometer	Vicinity of current ATP-3	slug test	23.8−60	2.1	
D & M 1982	TH-26	piezometer	Vicinity of current ATP-3	slug test	23.8−60	1.9	

Table 3–13. Summary of Historical Results of Hydraulic Parameters for the Alluvial Aquifer (continued)

Source	Location	Piezometer Well/Boring	Location Description	Test Type	Depth (bgs)	K_h (ft/day)	K_v (ft/day)
ORNL 1998	ATP-2-S	well	Between southeast base of the pile and river	aq test	28 - 38	22	
SMI 2001	SMI-PZ-1M	well	SMI-PW01 cluster	aq test		136.1	19.1
SMI 2001	SMI-PZ-1M	well	SMI-PW01 cluster	aq rec test		93.1	
SMI 2001	SMI-PZ-2M-1	well	SMI-PW02 cluster	aq test		185	29.6
SMI 2001	SMI-PZ-2M-1	well	SMI-PW02 cluster	aq rec test		202	
SMI 2001	SMI-PZ-2M-2	well	SMI-PW02 cluster	aq test		181.3	18.1
SMI 2001	SMI-PZ-3D	well	SMI-PW03 cluster	aq test		101.3	9.7
SMI 2001	SMI-PZ-3D	well	SMI-PW03 cluster	aq rec test		181.6	
SMI 2001	AMM-2	well	Between PW-01 and PW-02	slug test		80.2	
SMI 2001	ATP-2-S	well	Between southeast base of the pile and river	slug test		69.6	
SMI 2001	ATP-3	well	North of pile	slug test		2.1	
SMI 2001	AMM-3	well	Off southern tip of pile	slug test		38.3	
SMI 2001	MW-3	well	Vicinity of current SMI-PW01	slug test		49.4	
SMI 2001	SMI-MW-01	well	Northeast of pile, along river	slug test		45.5	
SMI 2001	SMI-PZ-1S	well	SMI-PW01 cluster	slug test		21.7	
SMI 2001	SMI-PZ-3S	well	SMI-PW03 cluster	slug test		19.4	

Notes:
D & M 1975 refers to Dames & Moore 1975 Report
D & M 1981 refers to Dames & Moore 1981 Report
D & M 1982 refers to Dames & Moore 1982 Report
ORNL 1998 refers to the Oak Ridge National Laboratory 1998 Report
SMI 2001 refers to the Shepard Miller 2001 Report
bgs = below ground surface
lab perm = laboratory permeability test
field perm = field permeability test
aq test = aquifer test
aq rec test = aquifer recover test
K_h = horizontal hydraulic conductivity
K_v = vertical hydraulic conductivity

Table 3–14. Results of the 1960 Surface Water Sampling Near the Moab Millsite[a]

	Sampling Event	M-4[b] (Colorado River)	M-5[c] (Colorado River)	M-7[d] (Ditch Effluent)
Manganese (mg/L)	1	0.24	< 0.15	0.90
	2	0.15	0.28	< 0.15
Iron (mg/L)	1	0.08	0.06	0.68
	2	0.08	0.08	0.52
Arsenic (mg/L)	1	< 0.01	< 0.01	1.3
	2	< 0.01	< 0.01	< 0.01
Selenium (mg/L)	1	0.006	0.001	< 0.001
	2	< 0.001	0.001	< 0.001
Vanadium (mg/L)	1	< 0.005	0.060	54.5
	2	0.015	0.080	37.6
Copper (mg/L)	1	0.007	0.005	0.015
	2	0.008	0.003	0.005
Sodium (mg/L)	1	260	260	-
	2	280	254	-
Fluoride (mg/L)	1	0.64	0.62	-
	2	0.55	0.62	2.0
Chloride (mg/L)	1	94	102	388
	2	108	109	358
Nitrate as N (mg/L)	1	1.4	1.7	16.4
	2	1.5	2.0	16.4
Sulfate (mg/L)	1	826	832	2,600
	2	871	916	2,460
Dissolved solids (mg/L)	1	1,810	1,800	8,470
	2	1,860	1,890	7,830
Total hardness[e] (mg/L)	1	840	811	264
	2	849	840	217
Total alkalinity[e] (mg/L)	1	146	142	1,820
	2	154	150	1,860
pH (s.u.)	1	7.9	8.0	9.8
	2	7.9	7.9	10.0

[a] Study conducted by U.S. Department of Health, Education, and Welfare.
[b] 0.6 mile upstream from effluent discharge point.
[c] 2 miles downstream from effluent discharge point.
[d] Sample collected in ditch before effluent discharge to river.
[e] $CaCO_3$ equivalent.

Constituents with concentrations that were elevated in the effluent (compared to the upstream location) were arsenic, vanadium, fluoride, sulfate, and TDS, among others. However, by the time the river water reached the first downstream location, concentrations of all chemical constituents except vanadium appear indistinguishable from those from the upstream sampling point. The report also noted small increases in radioactivity above background, but indicated that these were of minor public health significance. These results are some of the few data that were obtained for site surface water when process effluent was actively being discharged to the Colorado River. Most subsequent surface water monitoring occurred in the later operational and post-operation years when direct discharge to the river had ceased.

3.9.3.1 Surface Water Monitoring

As noted by Canonie (1994), surface water monitoring of the Colorado River began in 1981 with one upgradient and five downgradient sampling stations. In 1984, all but the upstream station (SWAM) and the station ¼ mile downstream (SW-1) of the mill were dropped. From 1981 to 1984 sampling was quarterly; in 1984 sampling was either quarterly or yearly depending on the constituent (Canonie 1994). Constituents analyzed were a subset of the ground water constituent list. As with ground water, some constituents were eliminated from further analysis in 1984 (Canonie 1994). It appears that regular monitoring was replaced by characterization completed for specific studies sometime in the mid-1990s.

3.9.4 Aquatic Studies

Because of concerns raised in its biological opinion regarding effects of ammonia on aquatic life in the Colorado River, particularly endangered fish, the USF&WS requested that USGS conduct research to determine potential impacts of the tailings pile on endangered fish. The Columbia Environmental Research Center, Biological Resources Division, began a series of studies in 1998, which were completed in 2000. The studies included contaminant mapping by sampling and analysis of soil pore water and adjacent nearshore river samples, various types of toxicity testing, and a benthic invertebrate assessment.

Toxicity testing was performed using larval and juvenile fish; Colorado pikeminnow, razorback suckers, and fathead minnows were used in various tests, depending on availability. Both laboratory and in situ tests were performed, and different parameters were varied during different tests. Results were compared between tests using different water types, pH, temperature, and ammonia concentrations.

Results of the toxicity testing indicated that the endangered species are not more sensitive to contaminants than standard test species, and actually appeared to be less sensitive. It was further concluded that Utah chronic surface water standards for ammonia that existed at the time would be protective of individual endangered fish by a factor greater than two (Fairchild et al. 2002). Also, the analysis of data from the benthic invertebrate survey indicated that ammonia contamination had no effect on the benthic invertebrate community. Concentrations of some metals (e.g., copper, manganese, zinc) and radionuclides were elevated in some areas but did not approach levels of concern (Fairchild et al. 2002). No specific recommendation was made regarding a safe level of ammonia in the river.

3.10 Transition to DOE

In October 2000, Congress passed the Floyd D. Spence National Defense Authorization Act for Fiscal Year 2001, and subsequently the title and cleanup responsibilities for the site were transferred to DOE. As required by the act, DOE prepared a Plan for Remediation (DOE 2001a) to evaluate the costs, risks, and benefits associated with reclamation on site versus relocation of the pile to an alternative location. This plan was based on an analysis of existing data and results of studies previously prepared for the site. DOE made no recommendation for tailings disposal as a result of this analysis, but several issues were identified that required further clarification or investigation to better support a recommendation. Most of these centered on apparent inconsistencies in the existing conceptual model for the site or the lack of conclusive data to support the interpretation of key aspects of site geology or hydrogeology. For example, it was known from previous work that a subsurface brine zone was present in at least part of the site area. However, little was known regarding the flow path and discharge zone, the extent, or geometry of the brine. The presence of a brine zone has implications for aquifer classification and for the design of an extraction system to recover site-related constituents. The source of freshwater recharge to the aquifer was also in question, and it was unclear whether this recharge could provide a source of usable water in the vicinity of the site. It had been speculated that a significant amount of freshwater recharge to the alluvial system was supplied by underlying bedrock units east of the Moab Fault (SMI 2001), but understanding of the subsurface geology was somewhat limited.

Interactions between the tailings pile and underlying alluvial system were also somewhat unclear. Limited ground water data were available for the area immediately underlying the pile. The pile was assumed to be the major source of ammonia contamination at the site; however, ground water collected from beneath the pile appeared to have minimal contamination. In addition, few data were available regarding potential soil contamination beneath the pile and whether contaminated soil could represent an ongoing source of ground water contamination. Due to uncertainties regarding tailings seepage rates, the extent of a dense slime layer at the base of the pile, and ammonia concentrations in the pore fluid, it was also unclear whether the pile and seepage from it could have an impact on the ability to remediate ground water contamination. DOE proposed to conduct additional activities and analyses to address these uncertainties. Section 4.0 describes the result of that additional work.

As required by the Floyd D. Spence Act, the National Academy of Sciences reviewed the Draft Preliminary Plan for Remediation (PFR) and came up with their own assessment of that plan and identified information that was needed to make a decision regarding disposal of the Moab mill tailings (NAS 2002). Issues raised by the NAS that are pertinent to the scope of the SOWP included those identified by DOE. NAS also raised concerns regarding long-term integrity of the pile in a dynamic geologic setting, potential effects of rewetting the base of the pile should a flood event occur, potential for construction water to remobilize constituents if the pile was capped or relocated, the actual suitability of habitat adjacent to the site for threatened and endangered fish, and the feasibility of designing a cover for the pile that would be adequately protective of ground water. NAS also identified other issues, such as consequences of catastrophic failure of the pile, potential impacts of global warming on cell performance, and socioeconomic considerations—issues beyond the scope of this document.

After DOE completed the Plan for Remediation and NAS reviewed the document, DOE-Headquarters made the decision that the PFR would not be revised and that, instead, DOE would proceed to an EIS to support its decision-making for remediation of the Moab site. The NEPA process is ongoing and the EIS is in preparation. This SOWP provides the technical basis for selection of an appropriate ground water compliance strategy for the Moab site, and results of this analysis are included in the EIS.

End of current text

4.0 DOE Field Investigations

This section describes field investigations conducted by DOE after assuming responsibility for the Moab site. Some activities were performed to support selection of the long-term remediation approach at the site and to resolve issues resulting from the PFR. Other activities were performed to support more near-term actions focused on protection of important habitat for endangered fish. All of these activities are described in this section. Results of these DOE investigations are integrated with relevant results of previous studies (Section 3) to develop the most current understanding of the natural systems at the site (Section 5). All data are synthesized into the site conceptual model presented in Section 6.

As noted previously, in developing the targeted ground water remediation alternatives presented in the draft PFR (DOE 2001a), DOE did not perform any characterization or modeling activities. The information presented in the PFR was extracted from existing documents, in particular, the *Infiltration, Seepage, and Groundwater Contamination Modeling for the Atlas Mill Tailings Pile* (CNWRA 1998), *Limited Groundwater Investigation of the Atlas Corporation Moab Mill* (ORNL 1998a), and the Moab trustee reports *Tailings Geochemistry Atlas Mill Site* (SRK 2001) and *Site Hydrogeologic and Geochemical Characterization and Alternatives Assessment for the Moab Uranium Mill Tailings Site* (SMI 2001). As discussed in Section 3.10, results of some of these earlier studies were somewhat ambiguous or seemingly contradictory. Both DOE and NAS identified additional subsurface hydrogeologic and chemical characterization needed to validate and refine the conceptual site model and reduce uncertainties in the ground water remediation alternatives.

DOE collected site characterization data in 2002 to address the issues identified in the draft PFR and NAS comments. Specifically, data were collected to better define the location and geometry of the brine at the site, to better understand freshwater recharge and the water budget, and to evaluate the role of the tailings pile and underlying soils as a source of ground water contamination. All fieldwork and data quality objectives applied to these data collection activities were performed in accordance with the *Work Plan for Groundwater and Tailings Pile Characterization Activities to Support the Plan for Remediation* (DOE 2002j). Sequencing the activities to achieve a more logical data collection strategy optimized the field investigations. The 2002 field investigations were sequenced as follows:

1. Collecting in situ soil conductivity measurements to delineate the brine zone.
2. Core sampling and installation of permanent alluvial and bedrock monitor wells; locations selected to better understand bedrock/alluvium relationship and effect of tailings pile on ground water.
3. Surveying the elevations and location coordinates of wells to provide a consistent datum.
4. Sampling new and key existing monitor wells to provide a current comprehensive set of baseline ground water quality data; selected wells were sampled at discrete intervals to define vertical chemical gradients and help delineate the brine zone.
5. Baseline surface water sampling to provide a current comprehensive set of surface water quality data.

6. Laboratory analysis of subpile soil samples to evaluate the impacts of a potential continuing source of ground water contamination that would limit the effectiveness of a remediation strategy.

Information obtained from each of the above activities was integrated with existing data to revise the site conceptual model and to refine the data collection needs. This integration was performed either concurrently with or before proceeding to the next characterization activity.

Additional field activities were conducted in 2001 and 2002 to support DOE initial and interim actions at the site. Results of previous investigations (ORNL 1998a and SMI 2001) indicated that site-related contaminants have leached from the tailings pile into the shallow ground water and that some of the more mobile constituents have migrated downgradient and are discharging to the Colorado River adjacent to the site. The initial and interim actions are designed as temporary measures to reduce the ecological risk to sensitive aquatic species that may inhabit the slow-moving water (backwaters) at the edge of the river where elevated ammonia concentrations have been observed. (Note that the term “backwater” is used here to denote any slow-moving waters. A “true” backwater as used by aquatic biologists, has a much more specific definition. See Section 5.6.8 for further discussion.)

These temporary actions would continue until a final long-term ground water remediation alternative is selected in the EIS. The initial action consists of applying uncontaminated river water to dilute ammonia concentrations in the backwaters; however, this action has not yet been implemented due to the recent low river flows and the lack of backwater areas. The interim action consists of ten extraction wells located along the bank of the Colorado River to intercept the center of the ammonia plume before it discharges to the river. The effluent is pumped into an evaporation pond located on top of the tailings pile. Construction of the interim action was completed in September 2003, and over one million gallons of plume water was treated during the first month of operations in October 2003.

Data collection activities performed in 2001 and 2002 specific to the initial and interim actions consist of mapping different morphological features of the river adjacent to the site, determining the distribution of ammonia in the shallow ground water near the backwater areas, mapping vegetation types and determining transpiration rates, installing a stilling well along the river to measure daily changes in the river elevation, and aquifer testing to evaluate the effect of upconing from the underlying brine unit into the freshwater system and to refine hydraulic parameters for the flow model. All fieldwork and data quality objectives applied to the data collection activities for these temporary actions were performed in accordance with the *Work Plan for Characterization to Support Groundwater Immediate and Interim Remediation Activities* (DOE 2001b).

Throughout this section and the rest of this document, concentrations of ammonia are cited as total ammonia (all species) reported as nitrogen (N) unless otherwise specified.

4.1 Soil Conductivity Measurements

Electrical soil conductivity logs were collected by a direct-push method to map the vertical and horizontal extent of the brine zone. Soil conductivity measurements were also performed to characterize the saltwater interface (i.e., sharp or diffuse), if observed, between the upper freshwater zone and the lower brine zone. The presence of a brine unit across the site would preclude upward flow of freshwater from the underlying bedrock formation. The presence of a naturally occurring brine unit could classify the ground water as limited use and would qualify the ground water for supplemental standards based on concentrations of TDS in excess of 10,000 mg/L [40 CFR 192.11(e)(1)].

A summary of the soil conductivity measurements performed at the Moab site from June 21 through July 13, 2002, is presented in the following sections. Details are presented in the *Soil Conductivity Investigation Results* (Appendix D, Calculation X0022900).

4.1.1 Soil Conductivity Measurement Locations

The eleven locations (MOA-358, and MOA-360 to -369) shown in Figure 4–1 were probed by ConeTec, Inc., using a Geoprobe Systems, Inc., SC400 soil conductivity tool. A Marl (M5T) Rhino rig was used to advance the probe into the subsurface at each measurement location. Location coordinates and ground elevation for each measurement location were surveyed using a survey grade portable global positioning system (Trimble Model 4700 receiver and TSC–1 data logger). A summary of the survey coordinates, ground elevations, and data collected at each location is presented in Table 4–1.

Table 4–1. Electrical Soil Conductivity Measurement Locations

Location ID (MOA)	Northing	Easting	Ground Elevation (ft)	Total Depth of Probe (ft bgs)	Date of Measurement	Comment
0358	6664481	2186199	3,966.6	90.6	6/21/2002	Adjacent to SMI-PW-01 well cluster
0360	6666201	2183223	4,001.1	38.7	7/11/2002	
0361	6666620	2183760	3,999.8	68.4	7/11/2002	
0362	6667191	2186378	3,987.7	45.5	7/13/2002	Water collected at 39 and 55 ft bgs
0363	6667106	2186749	3,973.3	31.6	7/12/2002	
0364	6666789	2187326	3,963.9	63.25	7/11/2002	Water collected at 40 and 54 ft bgs
0365	6666339	2187333	3,967.6	33.75	7/12/2002	
0366	6666790	2184978	3,989.9	45.55	7/13/2002	
0367	6666320	2185229	3,982.5	75.75	7/9/2002	Duplicate measurement to 45 ft
0368	6665946	2185950	3,963.5	31.3	7/11/2002	
0369	6665686	2186798	3,964.9	33.45	7/13/2002	

Notes: Coordinates are Modified Utah State Plane, Central Zone, North American Datum (NAD) 1983/1994.
Ground elevations are North American Vertical Datum (NAV) 1988.
ft bgs = feet below ground surface.

4.1.2 Soil Conductivity Profiles

Electrical conductivity results as a function of depth at each test location is provided in Figure 4–2, Figure 4–3, and Figure 4–4. The conductivity profiles shown are plotted at 2,000 millisiemens per meter (mS/m) full scale. Profiles reflecting the saltwater interface are found at locations MOA-358, -362, and -364.

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Figure 4–1. Soil Conductivity Measurement Locations

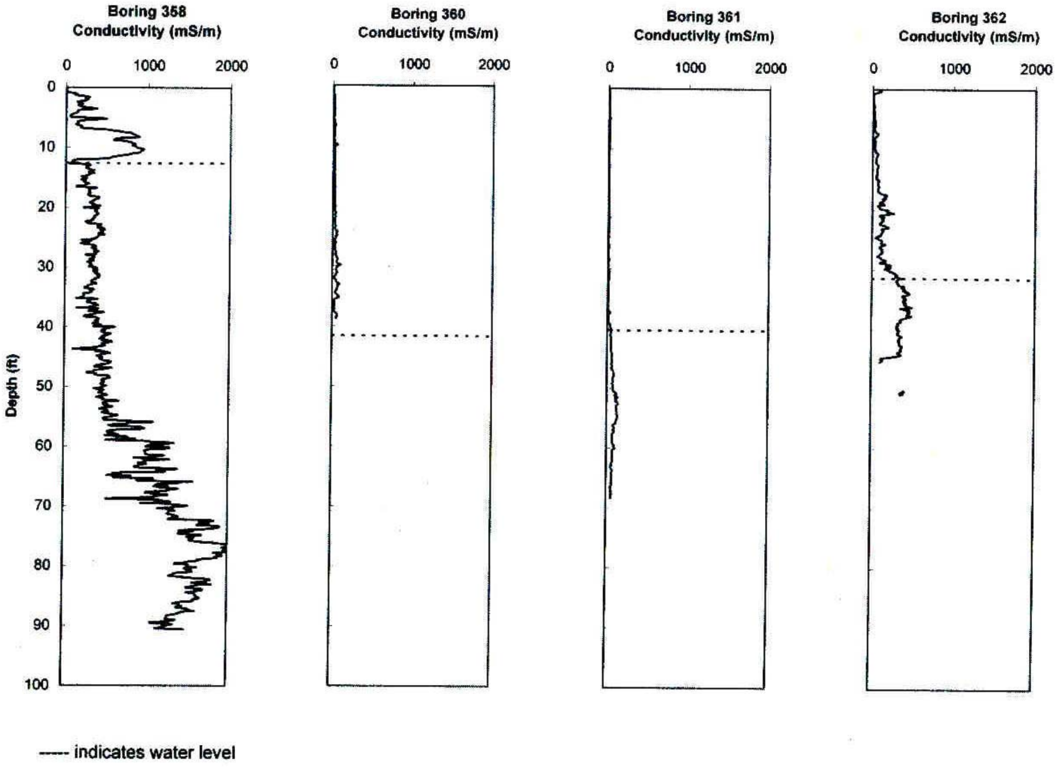

Figure 4–2. Soil Conductivity Profiles for Locations MOA-358, -360, -361, and -362

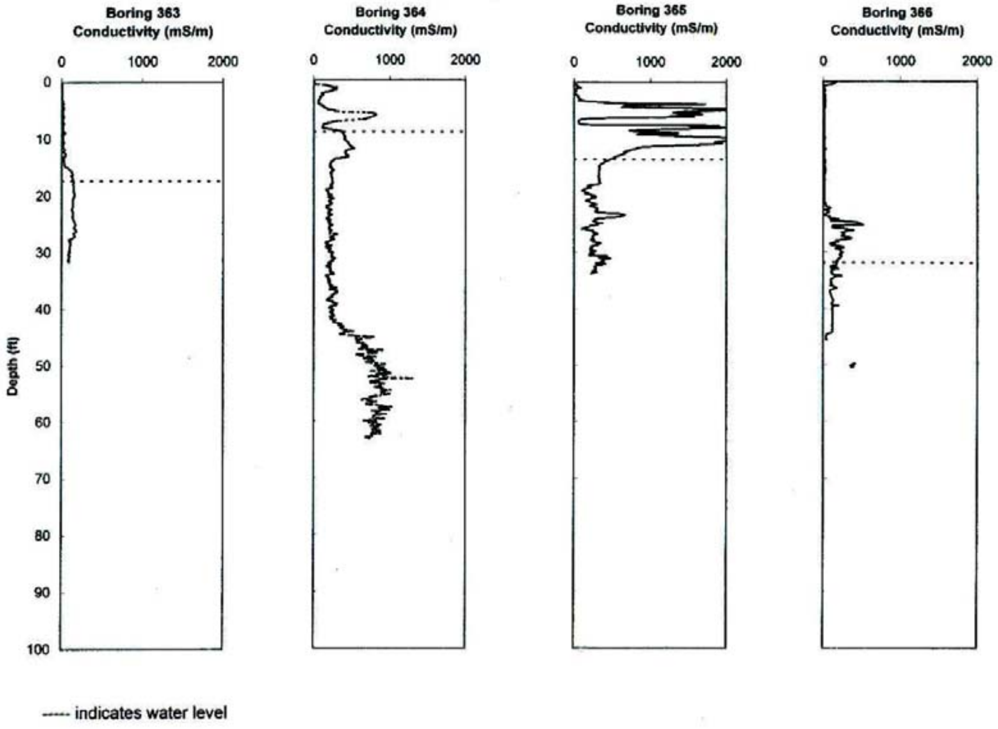

Figure 4–3. Soil Conductivity Profiles for Locations MOA-363, -364, -365, and -366

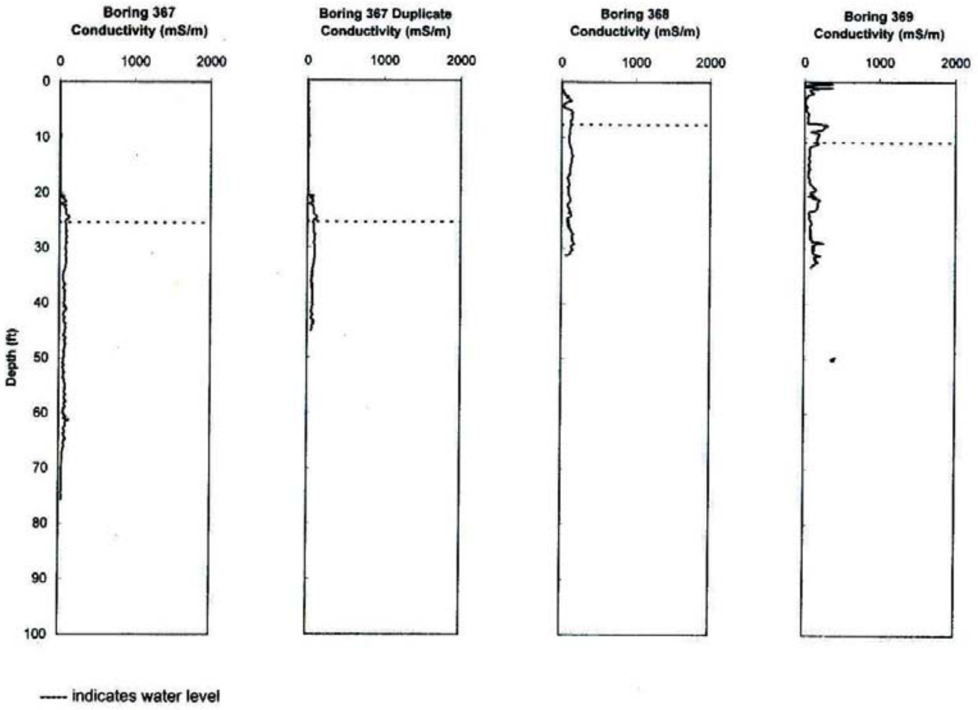

Figure 4–4. Soil Conductivity Profiles for Locations MOA-367, -367 (duplicate), -368, and -369

4.1.3 Reproducibility of the Soil Conductivity Method

Reproducibility of the soil conductivity method was evaluated in the field by performing a duplicate measurement at test location MOA-367 to 45 ft. However, a true duplicate measurement could not be performed at exactly the same location because the conductivity probe needs to be in direct contact with the soil. Probing the first hole a second time to obtain the duplicate measurement would result in an inadequate contact between the hole and the conductivity probe. Therefore, the second measurement location was offset a few feet from the first measurement location. Comparisons between the first measurement (Test 1) and the duplicate measurement (Test 2) are presented in Figure 4–5 and Figure 4–6. Excellent reproducibility in the method is evidenced by the high coefficient of determination (r^2) value of 0.91 shown in the regression equation.

4.1.4 Comparison of Soil Conductivity versus Ground Water Quality

Electrical soil conductivity measurements at test location MOA-358 were performed adjacent to (approximately 11 ft from) the SMI-PW-01 well cluster. TDS analyses on ground water grab samples collected at discrete depth intervals from the SMI-PW-01 well cluster previously reported in *Characterization of Groundwater Brine Zones at the Moab Site* (Appendix D, Calculation X0013800) provide a basis for comparison to the electrical soil conductivity results obtained at test location MOA-358. TDS concentrations in the ground water grab samples and the soil conductivity measured from approximately the same depth interval are summarized in Table 4–2. A linear regression analysis and a comparison between the TDS concentrations and the soil conductivity results at location MOA-358 are shown in Figure 4–7 and Figure 4–8, respectively.

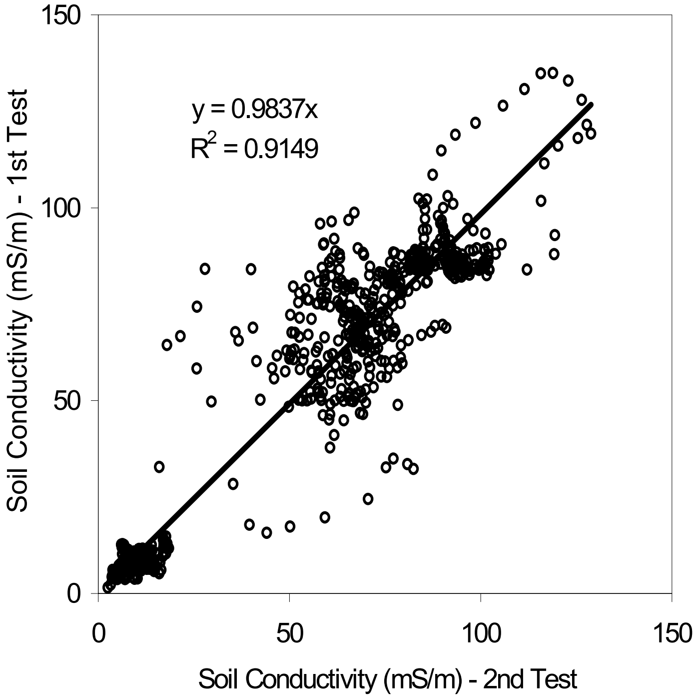

Figure 4–5. Regression Analysis of Duplicate Measurement Results at Location MOA-367

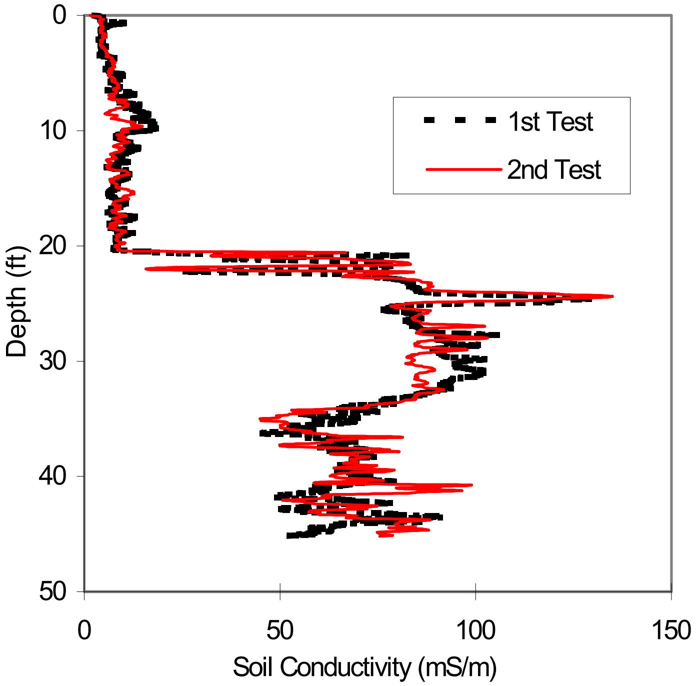

Figure 4–6. Comparison of Duplicate Measurement Results at Location MOA-367

Table 4–2. Soil Conductivity and TDS Results at Discrete Depth Intervals

Location ID (MOA)	Depth (ft)	Soil Conductivity (mS/m)	TDS in Water (mg/L)
358	19	322.8	12,325
358	23	406.73	12,400
358	28	339.87	12,100
358	33	285.03	13,525
358	38	292.33	15,113
358	43	456.59	18,120
358	48	390.07	18,540
358	53	538.89	20,580
358	58	641.58	46,100
358	60	1,095.37	34,433
358	62	989.13	46,350
358	75	1,560.84	77,600
362	39	335.75	3,480
362	55	NA	5,567
364	40	209.27	7,910
364	54	802.27	19,220

TDS in water samples were collected from SMI-PW–01 located adjacent to MOA–358.
Depth is feet below ground surface.
NA—soil conductivity measurement was not performed at this depth interval.

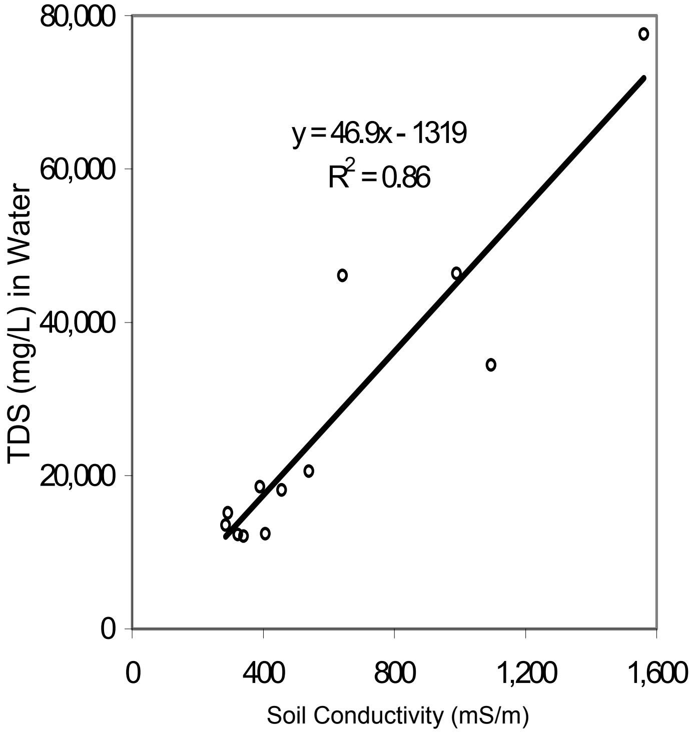

Figure 4–7. Regression Analysis of TDS Concentrations in Ground Water Samples versus Soil Conductivity Results at Location MOA−358

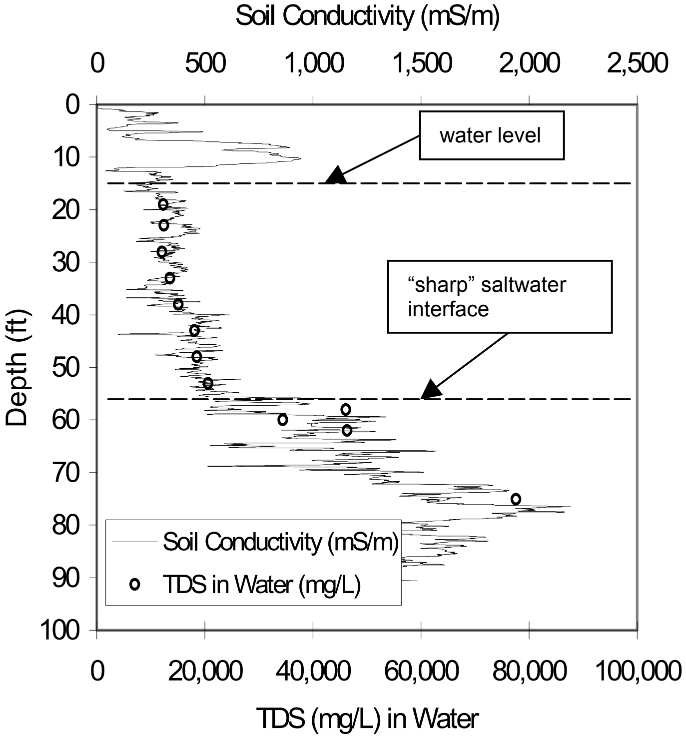

Figure 4–8. Comparison of TDS in Ground Water Samples at Discrete Depth Intervals versus Soil Conductivity Results at Location MOA−358

Good agreement is obtained between the electrical soil conductivity results and TDS as indicated by the relatively high coefficient of determination (r^2) value shown in the regression equation presented in Figure 4–7. Similar agreement is obtained for specific conductance, density, ammonia, and chloride in ground water. The *Soil Conductivity Investigation Results* (Appendix D, Calculation X0022900) presents the results of these regression analyses. Conversely, a relatively poor linear correlation is obtained between soil conductivity and sulfate in ground water (Appendix D, Calculation X0022900).

A sharp increase in soil conductivity at approximately 55 ft below ground level occurs in the electrical conductivity profile obtained at location MOA-358 (Figure 4–8). At depths less than 55 ft, soil conductivity in the saturated zone ranges between approximately 400 and 500 mS/m. These relatively low soil conductivity values correspond with TDS concentrations that range between 10,000 and 20,000 milligrams per liter (mg/L) in ground water. At approximately 55 ft in depth, the soil conductivity increases from approximately 500 to over 1,000 mS/m. This increase in soil conductivity corresponds with an increase in TDS concentration that ranges from approximately 20,000 to 40,000 mg/L in ground water. The highest soil conductivity value (greater than 2,000 mS/m) is observed at a depth of approximately 80 ft and corresponds with the highest TDS concentration (approximately 80,000 mg/L) measured in ground water at location MOA-358.

The pattern revealed by the sharp increase in conductivity at location MOA-358 is similar to the conductivity profile observed for test location MOA-364 at a depth of approximately 45 ft (see the 2,000 mS/m full-scale plot, Figure 4–3). At a depth of 45 ft the conductivity increases sharply from approximately 300 to 1,000 mS/m, suggesting the presence of a contact between the upper freshwater (<10,000 mg/L TDS) and deeper more saline waters (>10,000 mg/L TDS). Ground water grab samples collected at depths of 40 and 54 ft (Table 4–2) indicate TDS concentrations of 7,910 and 19,220 mg/L, respectively, verifying the presence of the contact.

A slight increase in soil conductivity is evident in the profile developed for test location 362 at a depth of approximately 30 ft. This increase in conductivity is not as pronounced as that observed for test locations MOA-358 and -364 (see the 2,000 mS/m full-scale plot, Figure 4–2), suggesting the TDS concentration is less than 10,000 mg/L at depth. Ground water grab samples collected at test location MOA-362, from depths of 39 and 55 ft, indicate TDS values of 3,480 and 5,567 mg/L, respectively, verifying the presence of relatively low TDS concentrations (Table 4–2). Similarly, the soil conductivity profiles for the other test locations (borings MOA-361, -363, -365, -366, -367, -368, and -369) suggest that relatively low TDS concentrations (<10,000 mg/L) are present in the ground water. Probe depths at these locations range from 31.30 ft to a maximum of 75.75 ft. Ground water was not reached at location MOA-360. Attempts to probe deeper were prevented by probe refusal in all cases.

4.2 Monitor Well Installations

Permanent and temporary monitor wells were installed to collect ground water samples for characterization of water quality and to provide a means to determine hydraulic properties of the alluvial and bedrock aquifers. Fourteen permanent monitor wells (2.0 inch diameter) were installed during the drilling campaign conducted at the Moab site from July 9, 2002, through August 25, 2002. Boart Longyear Company used a Gus Pech 300 Sonic Rig to perform the drilling for the permanent wells. Nine smaller diameter (1.0 inch) temporary wells were installed in December 2001, five were installed in May 2002, and three were installed in October 2002. All the small-diameter temporary wells were installed using a Geoprobe direct-push rig. The following procedures were used for monitor well installation:

- LQ–14(P), “Technical Comments on ASTM D 5092—Standard Practice for Design and Installation of Ground Water Monitor Wells in Aquifers.”
- GN–13(P), “Standard Practice for Equipment Decontamination.”

At the conclusion of the drilling campaign, the location coordinates and elevation for each monitor well were surveyed using a survey grade portable global positioning system (Trimble Model 4700 receiver and TSC-1 data logger). Lithologic logs, well construction information, and field sampling results for the permanent and temporary wells installed at the Moab site during 2001 and 2002 are summarized in the following sections. A comprehensive map showing all the existing monitor wells currently at the Moab site is provided in Plate 1.

4.2.1 Permanent Monitor Wells

A total of 14 permanent ground water monitor wells were installed in 2002 using the casing-advance sonic method. Eight wells were installed in the alluvial aquifer (MOA-436 through -440 and MOA-442 through -444) and six in the bedrock formation (MOA-430 through -435). Three of the monitor wells (MOA-430, -432, and -434) were installed in background areas north of the site to provide additional information for the characterization of upgradient ground water that is unaffected by the former milling operations. The remaining 11 were installed on-site to characterize the nature and extent of ground water contamination and to determine the hydrologic properties of the aquifers. Three of the on-site alluvial wells were installed in the upper section of the alluvial aquifer beneath the tailings pile (MOA-437, -438, and -439). The permanent monitor well locations are shown in Figure 4–9.

All the permanent ground water monitor wells were installed by the sonic drilling method that uses a dual line of drill rods to advance the borehole to the desired depth. The inner drill rod containing the drill bit is always advanced ahead of the outer rods. The outer rods are used as casing to prevent collapse of the borehole during well completion. After the borehole is advanced to the desired depth the inner rods are removed and a 2-inch, flush joint, threaded, PVC casing and 0.010 or 0.020 inch slot size PVC screen was installed. A medium-grained sand pack (10-20 sieve size) was placed in the annular space from the bottom of the borehole to a depth several feet above the top of the well screen. A fine-grained sand pack (20-40 size) was then placed to fill several feet of the annular space above the medium-grained sand. Both sand packs consist of clean quartz sand. A bentonite seal was tremied into place above the fine-grained sand pack followed by Enviroplug grout to fill the annular space above the bentonite seal to within 3 ft of ground surface. The well completion materials were compacted into the borehole annulus by applying a resonant vibration to the outer casing as it was extracted. Concrete was used to fill the remaining annular space to the ground surface and to install a 3-ft diameter well pad with a locking steel protective cover. Construction details for the permanent monitor well installations are shown on the individual well completion logs provided in Appendix A and summarized in Table 4–3.

Table 4–3. Construction Details for Permanent Water Wells Installed in 2002 at the Moab Site

ID (MOA)	Northing State Plane (ft)	Easting State Plane (ft)	Ground Elevation (ft)	Borehole Diameter (inches)	Top of Casing Elevation (ft)	Casing Diameter (inches)	Casing Length (ft)	Borehole Depth (ft)	Screen Depth (ft)	Screen Length (ft)	Zone of Completion
0430	6667757	2182244	4022.6	6	4022.1	2	105.8	113	96	10	BR
0431	6666522	2182943	4004.4	6	4007.04	2	101.94	106	89	10	BR
0432	6667040	2184809	4001.7	6	4001.47	2	60.07	60.3	50	10	BR
0433	6666773	2184863	3990.2	6	3989.99	2	104.09	106	94	10	BR
0434	6667455	2186665	3990.6	6	3990.21	2	84.91	85.3	75	10	BR
0435	6667026	2186798	3969.1	6	3971.67	2	183.87	181.3	171	10	BR
0436	6666105	2186197	3968.5	8	3970.8	2	207.6	205.3	195	10	AL
0437	6665399	2183803	4045.9	8	4048.25	2	102.65	250	90	10	AL
0438	6665241	2185010	4052	8	4054.22	2	121.52	120	109	10	AL
0439	6664189	2184731	4052.9	8	4055.27	2	122.67	304	110	10	AL
0440	6665301	2182825	4068.3	6	4070.71	2	121.71	120	109	10	AL
0442	6663696	2184113	4020.2	6	4022.78	2	68.88	66.3	61	5	AL
0443	6666507	2182943	4004.4	6	4006.72	2	82.62	82.5	70	10	AL
0444	6667025	2186808	3968.9	6	3970.99	2	122.39	120.3	110	10	AL

Notes: State Plane coordinates are Modified Utah State Plane, Central Zone, North American Datum (NAD) 1983/1994
Elevations are North American Vertical Datum (NAV) 1988
Depth is feet below ground surface
AL = alluvium; BR = bedrock formation, undifferentiated

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Figure 4–9. Location of Wells Installed in 2001 and 2002 at the Moab Site

4.2.2 Temporary Monitor Wells

Nine 1-inch diameter temporary wells were installed in December 2001 (MOA-401 through -409), five were installed in May 2002 (MOA-410 through -414), and three were installed in October 2002 (MOA-455 through -457). All the small-diameter temporary wells were installed using a Geoprobe direct-push rig. The nine wells installed in December 2001 are located along the riverbank adjacent to the tailings pile to investigate the ammonia concentration in the shallow alluvial aquifer that is potentially discharging to the backwater areas adjacent to the site. Five wells installed in May 2002 are located in areas of the site where organic solvents, such as kerosene, isodecanol, and di(2-ethylhexyl) phosphoric acid (see Section 3.0), may have been used in the milling operations and potentially contaminated the ground water. Three wells installed in October 2002 are each nested adjacent to a permanent bedrock or alluvial well to provide a shallow completion in the upper alluvial aquifer to collect vertical profile data. The three nested well locations consist of (1) MOA-433 and -455, (2) MOA-435, -444, and -457, and (3) MOA-434 and -456. The temporary monitor well locations are shown in Figure 4–9.

All the temporary alluvial wells were completed with a 1-inch, flush joint, threaded, PVC casing and 0.010-inch slot size PVC screen. Construction of all the temporary wells was similar to that of the permanent wells with the exception that some were constructed with only a fine-grained sand pack (20-40 size). Some of the temporary wells were also constructed with the bentonite seal extending to the ground surface. Construction details for the temporary monitor well installations are shown on the individual well completion logs provided in Appendix A and summarized in Table 4–4.

Table 4–4. Construction Details for Temporary Water Wells Installed in 2001 and 2002 at the Moab Site

ID (MOA)	Easting State Plane (ft)	Northing State Plane (ft)	Ground Elevation (ft)	Borehole Diameter (inches)	Top of Casing Elevation (ft)	Casing Diameter (inches)	Casing Length (ft)	Borehole Depth (ft)	Screen Depth (ft)	Screen Length (ft)	Zone of Completion
0401	6663842	2186101	3967.7	2	3969.60	1	20.06	19	13	4.9	AL
0402	6663682	2186089	3967.7	2	3968.63	1	19.53	19.5	13.43	4.92	AL
0403	6663535	2186078	3966.9	2	3968.95	1	20.55	19	13.26	4.92	AL
0404	6664017	2186184	3966.3	2	3968.30	1	20.45	19	13.28	4.92	AL
0405	6664404	2186331	3966.4	2	3968.47	1	22.36	21	15.12	4.92	AL
0406	6664631	2186333	3967.9	2	3969.91	1	20.3	19	13.12	4.92	AL
0407	6663348	2186030	3967.2	2.125	3969.09	1	20.39	19	13.33	4.92	AL
0408	6663836	2186100	3967.8	2.125	3969.17	1	29.57	30	23.03	4.92	AL
0409	6664220	2186249	3967	2.125	3969.03	1	20.53	19	13.33	4.92	AL
0410	6666393	2186368	3978.5	2	3979.11	1	24.58	24.5	18.8	4.92	AL
0411	6666026	2185885	3964.2	2	3964.88	1	9.5	9.5	3.54	4.92	AL
0412	6665724	2186705	3965	2	3965.76	1	13.61	12.85	7.56	4.92	AL
0413	6665542	2186159	3964.5	2	3965.33	1	13.58	12.75	7.48	4.92	AL
0414	6665294	2186493	3962.4	2	3963.20	1	10.65	9.85	3.57	5.92	AL
0455	6666763	2184863	3990.2	2.125	3990.20	1	48.5	50.5	43.35	4.9	AL
0456	6667450	2186667	3990.5	2.125	3990.46	1	55.41	57	50.3	4.9	AL
0457	6667019	2186801	3968.9	2.125	3971.30	1	33.39	32	25.84	4.9	AL

Notes: State Plane coordinates are Modified Utah State Plane, Central Zone, North American Datum (NAD) 1983/1994
Elevations are North American Vertical Datum (NAV) 1988
Depth is feet below ground surface
AL = alluvium

4.3 Water Sampling and Analysis

Ground water samples were collected during the field investigations using grab sampling techniques and low-flow methods. Grab samples were analyzed in the field for specific conductance and submitted to DOE's Environmental Sciences Laboratory (ESL) or Analytical Chemistry Laboratory; both are located at DOE's Grand Junction, Colorado, office. The samples were submitted for quick turnaround analyses so the site conceptual model could be updated as drilling and sampling progressed. Grab sampling results are provided in Appendix C.

Low-flow sampling techniques were used to collect ground water samples on a quarterly basis from existing monitor wells and from the new wells installed during the DOE field investigation. River water samples were collected concurrently with the quarterly ground water samples. All water samples collected quarterly were submitted to the Analytical Chemistry Laboratory for analysis. An independent data validation assessment was conducted on the analytical results of all the quarterly samples. Quarterly sampling results are provided in Appendix C.

A description of the sampling and analysis procedures used is presented in the following sections.

4.3.1 Water Sampling Procedures

Water sampling was performed in accordance with the *Sampling and Analysis Plan for the UMTRA Ground Water Project* (DOE 2002h) and the *Grand Junction Office Environmental Procedures Catalog* (GJO 6). The following specific procedures from the Environmental Procedures Catalog were used for water sampling:

- GT-1(P), "Standard Practice for Field Documentation Processes."
- GT-2(P), "Standard Practice for Sample Labeling."
- GT-3(P), "Standard Practice for Chain-of-Sample-Custody and Physical Security of Samples."
- LQ-2(T), "Standard Test Method for the Measurement of Water Levels in Ground Water Monitoring Wells."
- LQ-3(P), "Standard Practice for Purging Monitoring Wells."
- LQ-4(T), "Standard Test Method for the Field Measurement of pH."
- LQ-5(T), "Standard Test Method for the Field Measurement of Specific Conductance."
- LQ-6(T), "Standard Test Method for the Field Measurement of the Oxidation-Reduction Potential (Eh)."
- LQ-7(T), "Standard Test Method for the Field Measurement of Alkalinity."
- LQ-8(T), "Standard Test Method for the Field Measurement of Temperature."
- LQ-9(T), "Standard Test Method for the Field Measurement of Dissolved Oxygen."
- LQ-10(P), "Standard Practice for the Use of a Flow Cell for Field Measurements."
- LQ-24(T), "Standard Test Method for Turbidity in Water."

- LQ-11(P), "Standard Practice for the Sampling of Liquids."
- LQ-12(P), "Standard Practice for the Collection, Filtration, and Preservation of Liquid Samples."
- LQ-19(P), "Standard Practice for the Inspection and Maintenance of Ground water Monitoring Wells."

4.3.1.1 Grab Sampling

A casing-advance sonic drilling method was used to advance each borehole through the alluvium to the sampling zone of interest. Ground water grab samples were collected at discrete depths from selected borings using either the Hydropunch sampling method or bailer (first water only). One to two water samples were collected at each sampling zone to profile the water quality as a function of depth. Advancing the casing to the next deeper zone of interest allowed collection of multiple samples from the same borehole. All the deeper ground water grab samples were collected and analyzed in the same manner. Compared to the regular quarterly sampling described below, grab sampling procedures, timing, and analysis varied depending on the objectives of the sampling. In some cases, grab sampling was done to help guide additional field activities, to determine vertical stratification of constituents, or for some other purpose. Grab sampling and quarterly sampling results are presented separately in Appendix C.

4.3.1.2 Quarterly Sampling

Low-flow sampling was conducted between December 2001 and December 2002 for the five quarterly events listed in Table 4–5. Low-flow sampling is a method for collecting ground water samples using pump and flow rates of 100 to 500 mL per minute. The low flow rate avoids pulling stagnant water above the well screen into the sample line and avoids dislodging colloids sorbed to the aquifer grains.

Table 4–5. Quarterly Water Sampling Events Using Low-Flow Methods

Sampling Event	**Dates Sampled**	**Number of Locations**		
		Historical Wells	New Wells	Surface Water
1	December 2001	13	na	13
2	March 2002	12	na	14
3	May 2002	17	4	14
4	August and September 2002	20	20	14
5	December 2002	27	19	16

Samples were collected at selected historical wells, newly installed wells, and surface water locations. All quarterly samples collected during the 2001 and 2002 sampling events were submitted to the Analytical Chemistry Laboratory for analyses. Constituents analyzed during each sampling event are listed in Table 4–6.

Table 4–6. Analyses Performed For Quarterly Water Sampling Events

Analyte	Dec 2001	March 2002	May 2002	Aug/Sept 2002	Dec. 2002
Al				X	
Ammonia as N	X	X	X	X	X
Sb	X	X	X	X	X
As	X	X	X	X	X
B		X		X	X
Ba	X		X	X	X
Be				X	
Cd	X	X	X	X	X
Ca	X	X	X	X	
Cl	X	X	X	X	X
Cr	X	X	X	X	
Co			X	X	
Cu	X	X	X	X	
F				X	X
Gross Alpha	X	X	X	X	X
Gross Beta	X	X	X	X	X
Fe	X	X	X	X	X
Pb	X	X	X	X	
Pb-210	X	X	X	X	
Li				X	X
Mg	X	X	X	X	
Mn	X	X	X	X	X
Hg	X	X	X		
Mo	X	X	X	X	X
Ni	X	X	X	X	
NO_3	X	X	X	X	X
PCBs			X		
PO_4				X	
Po-210	X	X	X	X	
K	X	X	X	X	
Ra-226	X	X	X	X	X
Ra-228	X	X	X	X	X
Rn-222				X	
Se	X	X	X	X	X
Semivolatile organic compounds			X		
SiO_2				X	
Ag	X	X	X	X	
Na	X	X	X	X	X
Sr	X	X	X	X	X
SO_4	X	X	X	X	X
Tl	X	X	X	X	
Th-230	X	X	X	X	
Total organic carbon				X	
TDS	X	X	X	X	X
U	X	X	X	X	X
U-234, U 238			X	X	
V	X	X	X	X	X
Volatile organic compounds		X	X		
Zn	X	X	X	X	

4.3.2 Water Sample Analysis

Water samples submitted to the ESL were analyzed in accordance with relevant procedures as specified in the *Environmental Sciences Laboratory Procedures Manual* (GJO 210). Water samples submitted to the Analytical Chemistry Laboratory were analyzed as specified in relevant EPA guidelines or the *Handbook of Analytical and Sample-Preparation Procedures* (Lab-1, Lab-2, Lab-3, Lab-4). Accuracy was checked through internal laboratory quality-control checks, such as blind duplicates, splits, and known standards. A minimum of 10 percent of the samples collected and analyzed by the Analytical Chemistry Laboratory were field quality-control samples. Field quality-control samples included equipment rinsate blanks, trip blanks, check samples, and duplicates. These samples were submitted for the same analyses as the other field samples.

Analytical methods used by the Analytical Chemistry Laboratory for analysis of ground water and surface water samples are listed in Table 4–7. Sample preservation consisted of acidifying and storing the samples in an ice chest with Blue Ice (or equivalent) to cool during field sampling, packaging, and shipping. The list of analytes is based on an evaluation and screening of historical information that summarized water quality data for the Moab site (SMI 2001, NRC 1999, and ORNL 1998a). The most comprehensive baseline sampling and analysis was conducted in August/September 2002. Results from that sampling round were used to refine the analyte list for subsequent sampling. A description of the analyte screening and constituent selection process is presented in Appendix C. Sample handling, preparation, and analyses are described in the references shown in Table 4–7.

Final analytical results were entered into the SEEPro database, and an independent data validation was performed on the analytical results from the five sampling events listed in Table 4–5. (DOE 2002c, DOE 2002d, DOE 2002e, DOE 2002f, DOE 2003a, respectively). Results of the surface water and ground water analyses are included in Appendix C.

4.4 Water Elevation Measurements

Water levels were collected to determine ground water flow directions of the alluvial aquifer, horizontal and vertical gradients, and to establish the relationship between the alluvial aquifer and the Colorado River. These data were collected using a variety of methods. Surface water elevation data were collected from a stilling well established at a location in the Colorado River adjacent to the site. Ground water elevation data were collected using a hand operated water-level indicator and a series of electronic pressure transducers attached to data loggers installed at various locations across the site. Details regarding water elevation data collected from the stilling well and from ground water monitor wells are presented below.

4.4.1 River Water Elevation

River surface elevation data were collected to establish, among other things, a relationship between the river and the alluvial aquifer. In order to measure the actual river surface elevation, a stilling well (which included the installation of a pressure transducer and data logger) was established at the former millsite pumphouse (Figure 4–9) in February 2002.

Table 4–7. Analytical Chemistry Laboratory Sample Requirements and Analytical Methods

Analyte	Contract Required Detection Limit (mg/L)[a]	GJO Laboratory Technique[b]	EPA Analytical Method
Al	0.05	ICP-AES	SW-846 6010B
Ammonia as N	0.10	Colorimetric	EPA 350.1
Sb	0.003	ICP-MS, ICP-AES	SW-846 6020, SW-846 6010B
As	0.005	Hydride AA, GFAA, ICP-AES, ICP-MS	SW-846 7062, SW-846 6010B
B	0.10	ICP-AES	6010B
Ba	0.10	ICP-AES	SW-846 6010B
Be	0.003	ICP-AES	SW-846 6010B
Cd	0.001	ICP-MS, ICP-AES	SW-846 6020, SW-846 6010B
Ca	0.50	ICP-AES	SW-846 6010B
Cl	0.50	IC	SW-846 7199
Cr	0.01	ICP-AES, ICP-MS	SW-846 6020, SW-846 6010B
Co	0.05	ICP-AES, ICP-MS	SW-846 6020, SW-846 6010B
Cu	0.02	ICP-AES, ICP-MS	SW-846 6020, SW-846 6010B
F	0.10	IC	SW-846 7199
Gross Alpha	1.0	PC	SW-846 9310
Gross Beta	2.0	PC	SW-846 9310
Fe	0.03	ICP-AES	SW-846 6010B
Pb	0.003	ICP-MS, ICP-AES, GFAA	SW-846 6020, SW-846 6010B
Pb-210	1.5	LSc	N/A[c]
Li	0.0002	ICP-AES	SW-846 6010B
Mg	0.10	ICP-AES	SW-846 6010B
Mn	0.01	ICP-AES	SW-846 6010B
Hg	0.0002	CVAA	EPA 245.1 CLP-M, 245.5 CLP-M, 7470A, and 7471A
Mo	0.01	ICP-MS, ICP-AES	SW-846 6020, SW-846 6010B
Ni	0.04	ICP-AES, ICP-MS	SW-846 6020, SW-846 6010B
NO_3	1.0	IC	SW-846 7199
PCBs	0.001	GC	8082
PO_4	1.0 (0.10)	Colorimetric, IC	SW-846 7199
Po-210	1.0	AS	N/A[c]
K	0.10	ICP-AES, FAA	SW-846 6010B, SW-846 7770
Ra-226	1.0	AS, LSc	SW-846 9315, 9320, modified
Ra-228	1.0	BGC, LSc	SW-846 9315, 9320, modified
Rn-222	20	LSc	N/A[c]
Se	0.005	Hydride AA, GFAA, ICP-AES, ICP-MS	SW-846 6010B, SW-846 6020
Semivolatile organic compounds	--	GC-MS	SW-846 8270C
SiO_2	0.10	ICP-AES	SW-846 6010B
Ag	0.01	ICP-AES, ICP-MS	SW-846 6020, SW-846 6010B
Na	1.0	ICP-AES	SW-846 6010B
Sr	0.01	ICP-AES	SW-846 6010B
SO_4	1.0	IC	SW-846 7199
Tl	0.005	ICP-MS, GFAA, ICP-AES	SW-846 6010B, SW-846 6020
Th-230	1.0	ICP-MS, AS	SW-846 6020

Table 4–7. Analytical Chemistry Laboratory Sample Requirements and Analytical Methods (continued)

Analyte	Contract Required Detection Limit (mg/L)[a]	GJO Laboratory Technique[b]	EPA Analytical Method
Total organic carbon	0.3	IR	EPA 415.1
TDS	10	Gravimetry	Standards Methods 2540
U	0.001	ICP-MS	SW-846 6020
U-234, U 238	1.0	AS, ICP-MS	SW-846 6020
V	0.01	ICP-AES	SW-846 6010B
Volatile organic compounds	--	GC-MS	SW-846 8260B
Zn	0.01 (0.05)	ICP-AES, ICP-MS	SW-846 6010B, SW-846 6020

[a] Radionuclide detection limits are in units of pCi/L. The CRDL is typically set an order of magnitude less than the standards (see Appendix C for detailed explanation).

[b] The primary technique is listed first. Laboratory technique acronyms and abbreviations are defined below.

AS	Alpha Spectrometry
BGC	Beta/Gamma Coincidence
CVAA	Cold Vapor Atomic Absorption Spectroscopy
FAA	Flame Atomic Absorption Spectroscopy
GFAA	Graphite Furnace Atomic Absorption Spectroscopy
GC-MS	Gas Chromatograph - Mass Spectrometry
Hydride AA	Sodium Borohydride Reduction Atomic Absorption Spectrometry
IC	Ion Chromatography
ICP-AES	Inductively Coupled Plasma–Atomic Emission Spectrometry
ICP-MS	Inductively Coupled Plasma - Mass Spectrometry
IR	Infrared
LSc	Liquid Scintillation
PC	Proportional Counting

[c] N/A; no applicable EPA method.

According to the data collected at this location, the river elevation for a flow of 4,000 cfs is approximately 3,954 ft at the pumphouse. The peak elevation measured at the pumphouse stilling well from 2002 through 2003 was 3,962.5 in response to the 2003 spring runoff that had a daily mean peak of 26,200 cfs.

A series of four waterline surveys were completed between March and December 2002 to determine the Colorado River gradient adjacent to the site. Water elevations were measured using a survey grade portable global positioning system (Trimble Model 4700 receiver and TSC-1 data logger). Measurements were collected along the water's edge beginning at the boat landing north of the US-Highway 191 bridge and extending to a point approximately 3,500 ft south of the site boundary. Each survey was completed when the river flow was less than 4,000 cfs as result of the low flows experienced during 2002.

The river gradient was calculated from the northern end (in the vicinity of the former pumphouse) to the southern edge (to the river measuring point east of well TP-07) of the site. A steeper gradient is observed for the portion of the river adjacent to the southern half of the site (near well MOA-402 to the river measuring point east of well TP-07). Results of the river gradients measured for the entire survey and for the relatively steeper southern portion are presented in Table 4–8.

Table 4–8. Colorado River Gradients

Date	Colorado River Flow (cfs)[a]	River Gradient for Entire Waterline Survey (ft/ft)	River Gradient for Southern Waterline Survey (ft/ft)
March 11, 2002	2,360	0.00065	0.0018
May 20, 2002	3,280	0.00066	0.0018
August 12, 2002	1,840	0.00068	0.0025
December 4, 2002	2,460	0.00065	0.0021

[a] Flow measured at the USGS Cisco, Utah, gaging station

Results of the waterline surveys indicate that for flows less than 4,000 cfs, the gradient across the site ranges from 0.00065 to 0.00068 ft/ft. Adjacent to the southern half of the tailings pile the gradient ranges from 0.0018 to 0.0025 ft/ft.

Streamflow of the Colorado River has been collected by the USGS at gaging station 09180500 near Cisco, Utah, since 1914. This station, located approximately 35 miles upstream of the site, is the closest station and provides the most complete data set representing river flow at the Moab site. The USGS reports Colorado River streamflow data collected at the gaging station as daily mean flow in cubic feet per second. A comparison of the river elevation measured at the stilling well and the streamflow reported by USGS for the Cisco gaging station from February 2002 through June 2003 is shown in Figure 4–10. The data presented in Figure 4–10 show that fluctuations in the Colorado River streamflow at the Cisco station coincide with a similar response in the river elevation measured by the pressure transducer at the stilling well.

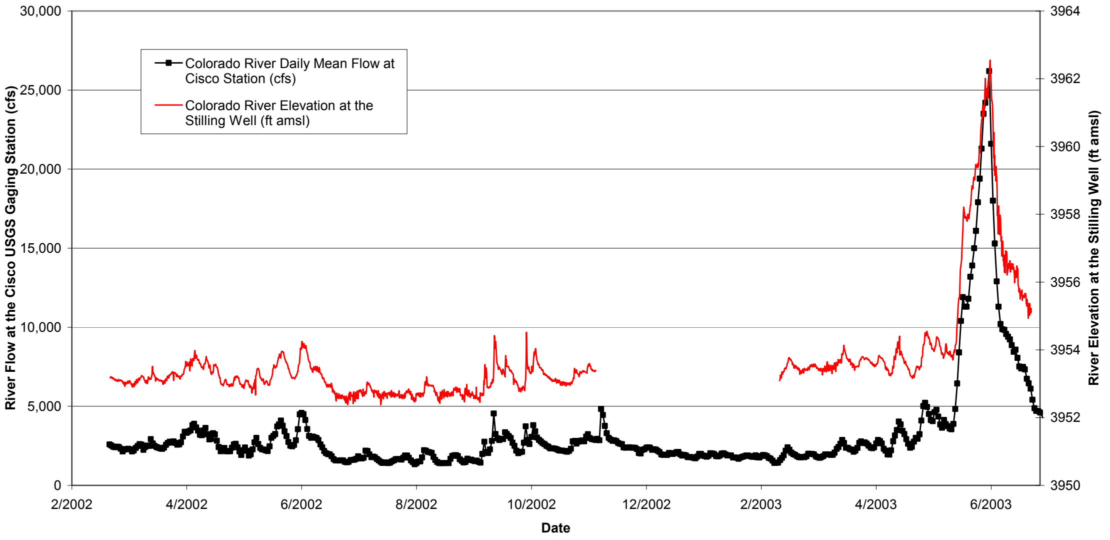

Figure 4–10. Comparison of the Daily Mean Flow Measured at the USGS Cisco Gaging Station and the River Surface Elevation Adjacent to the Site

River elevation and flow data presented in Figure 4–10 include the 2002 and 2003 spring runoff periods of the Colorado River. The maximum daily mean streamflow recorded during 2002 at the Cisco station was only 4,580 cfs (2002 was an extreme drought year), and the 2003 maximum daily mean streamflow measured was 26,200 cfs. The average maximum daily mean streamflow from 1959 (when the regulation of upstream flows began) through 2003 was 27,200 cfs.

A linear regression analysis between the streamflow data and the stilling well river surface elevation data is presented in Figure 4–11. The relatively high coefficient of determination (r^2) value of 0.97 suggests (assuming a similar shaped river channel) that the Colorado River streamflow can be used to estimate the river surface elevations at the stilling well location. The regression equation included in Figure 4–11 provides a reasonable relationship for flows between approximately 1,500 and 25,500 cfs.

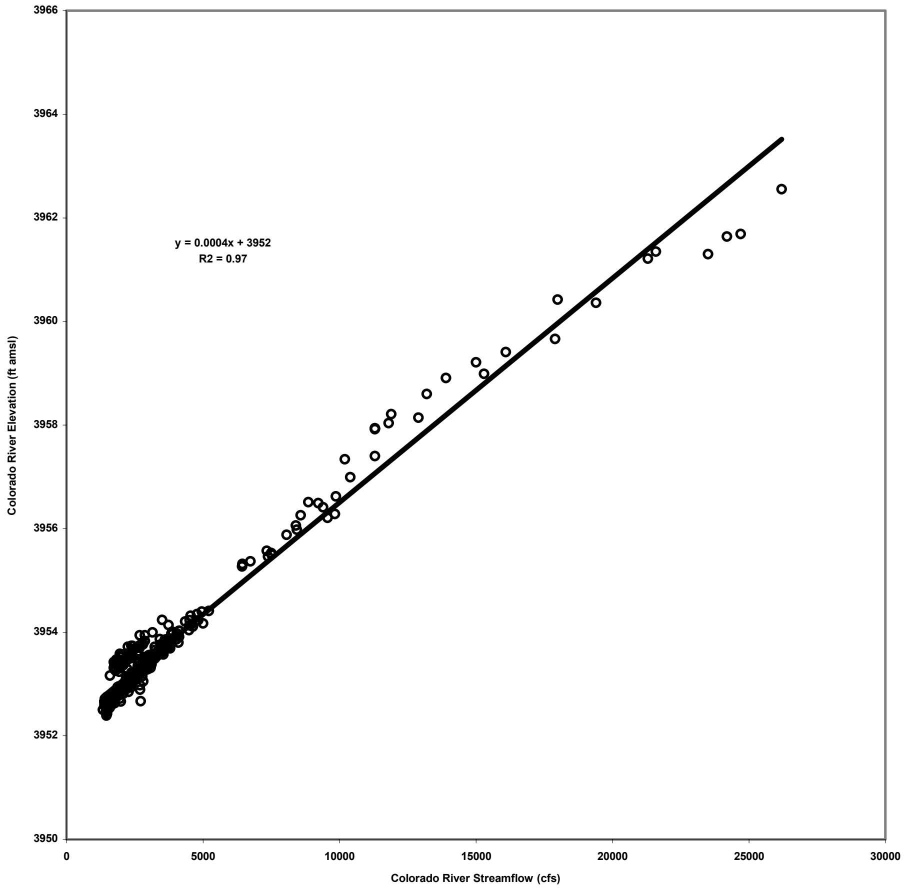

Figure 4–11. Linear-Regression Analysis Between River Surface Elevation and River Flow Rate

4.4.2 Ground Water Elevation

Ground water elevations tend to fluctuate in response to recharge events, river stage changes, and discharge events. Ground water elevation data have been measured by two methods: (1) hand measurements using an electronic sounder, and (2) pressure transducers attached to data loggers.

4.4.2.1 Hand Sounding Measurements

Ground water elevation data are typically collected using an electronic sounder. Data collected in this manner are generally associated with the quarterly ground water sampling events. Before a well is sampled, the depth to water is measured from the top of the well casing. The depth to water is entered into the database and converted into an elevation (feet above mean sea level) by subtracting the depth to water from the surveyed top of casing elevation. This data set is presented in Appendix B.

4.4.2.2 Pressure Transducer Measurements

Pressure transducers attached to data loggers are designed to continuously measure ground water fluctuations in the absence of site personnel. The instruments installed at the Moab site are programmed to measure the depth to water from the top of casing and the water temperature every 4 hours. These data are downloaded approximately every 4 months and loaded into the database. The depth to water is then converted to a ground water elevation (feet above mean sea level). Each time the data logger measurement is downloaded, the depth to water is measured by hand with an electric sounder to confirm the transducer is operating correctly. If the hand-measured water level is more than 0.1 ft different from the latest transducer measurement, the data logger is stopped, recalibrated using the correct water level, and restarted.

Eleven pressure transducers attached to data loggers were installed at the Moab site for long-term monitoring. Six other instruments were installed on site from February/March 2002 through August 2003. Monitor wells that have been equipped with pressure transducers, the month the instruments were installed and removed (if applicable), and the locations of the monitor wells are listed in Table 4–9. Time series plots of the water elevation measurements collected at each location listed in Table 4–9 are presented in Appendix B.

4.5 Piezometer Installations

Vibrating wire piezometers (VWP) were installed at three locations in the tailing pile to monitor pore pressure and gain an understanding of the rate of pore fluid seepage from the tailings to the ground water system. VWPs were also installed at three locations adjacent to the Colorado River to measure fluid pressure in the alluvial aquifer to evaluate ground water flow direction.

VWPs measure fluid pressures by monitoring the frequency of a vibrating wire that is attached to a pressure sensitive diaphragm. The diaphragm is open to the fluid and is protected from sediment with a 50-micrometer (µm) filter. A magnetic coil plucks the wire at a specified frequency and time to excite the wire. The wire vibrates at a resonant frequency dictated by the tension in the diaphragm, which depends on the fluid pressure. An increase in fluid pressure on the diaphragm will decrease the wire tension and thus the frequency of the vibrating wire. Conversely, a decrease in fluid pressure will increase wire tension and increase the resonant frequency of the vibrating wire.

Table 4–9. Summary of Pressure Transducers Installed at the Moab Site

Monitor Well	Month Installed	Month Removed	Location
SMI-PW01	Mar 02	NA	On site, between the tailings pile and the Colorado River
SMI-PZ1S	Mar 02	NA	On site, between the tailings pile and the Colorado River
SMI-PZ1M	Mar 02	NA	On site, between the tailings pile and the Colorado River
SMI-PZ1D	Mar 02	Nov 02	On site, between the tailings pile and the Colorado River
0449	Nov 02	NA	On site, within the PW01 cluster
SMI-PW02	Feb 02	NA	On site, between the tailings pile and the Colorado River
SMI-PZ2M1	Feb 02	Aug 03	On site, between the tailings pile and the Colorado River
SMI-PZ2M2	Feb 02	Aug 03	On site, between the tailings pile and the Colorado River
SMI-PZ2D	Feb 02	Aug 03	On site, between the tailings pile and the Colorado River
SMI-PW03	Mar 02	NA	On site, east of the tailings pile
SMI-PZ3S	Mar 02	Aug 03	On site, east of the tailings pile
SMI-PZ3M	Mar 02	Aug 03	On site, east of the tailings pile
SMI-PZ3D2	Mar 02	Aug 03	On site, east of the tailings pile
SMI-MW01	Mar 02	NA	On site, just off the Colorado River
0406	Mar 02	NA	On site, between the tailings pile and the Colorado River
0438	Nov 02	NA	On site, on top of the tailings pile
ATP-3	Nov 02	NA	On site, north of the tailings pile
RW-01	Nov 02	NA	Off site, upgradient (north) of the site

Notes: NA – Not applicable

Vibration of the wire is a function of the temperature. The temperature of the wire is measured by a thermistor located within the instrument to allow the reading to be compensated for temperature effects. Frequency readings from the VWPs are recorded using a Campbell Scientific CR10X data logger and controller equipped with an electronic interface that applies a series of frequencies to the wire and transmits the resonant frequency back to the data logger. The vibration frequency is converted by the data logger to a pressure through an empirical calibration of the relationship between the frequency and pressure.

Details of the installation, calibration, and results of the piezometers located in the tailings pile and the alluvial aquifer are presented in Appendix D, Calculations X0025700 and X0023000, respectively.

4.6 Core Sampling

A truck-mounted Gus Pech 300 Sonic Rig was used to collect core samples of unconsolidated sediment and tailings and samples of bedrock for lithologic logging and chemical analysis during installation of the permanent ground water monitor wells. Lithologic logs were prepared to support development of the site hydrogeologic model. Samples were collected for chemical analysis to determine distribution coefficients (K_d) and mobile fractions of site-related constituents in the subpile soils and column tests to aid in characterizing subsurface contaminant transport (Appendix D, Calculation X0023200 and X0023400).

The sonic drilling method uses a dual line of drill rods to advance the borehole to the desired sampling depth. The inner drill rod contains the drill bit and is also the core barrel. The drill bit is always advanced ahead of the outer rods. After the core barrel has been advanced approximately 5 ft, the outer casing is advanced to prevent collapse of the borehole and to seal it off from up-hole material that could contaminate the core sample. For added precaution to avoid cross-contamination when drilling through the tailings material, a third outer casing was set into a bentonite seal placed at the base of the tailings.

Continuous core samples were collected from the unconsolidated material and from the bedrock formation without the use of air or drilling fluids to ensure accurate, representative, and undiluted samples. Clean water was necessary as the circulation medium only in a few cases when drilling became difficult in the indurated bedrock formation. State-of-the industry coring practices were used to effect the highest core recovery possible, which in most cases was near 100 percent. Recovered core was vibrated out of the core barrel into plastic sheaths and placed in boxes within the longitudinal separators, from left to right, as a book would be read; that is, core was placed starting with the shallowest portion of the hole in the upper left corner and ending with core from the deepest portion of the hole in the lower right corner. Spacer blocks were inserted between the cored sections within the longitudinal separators where no recovery was noted. All core boxes, including the lids, were permanently marked showing top and bottom and the beginning and ending depths for the core, and stored in a core room at the Moab site. An index of the core stored at the site is presented in Appendix A by borehole, depth interval, and box number.

All sediment, soil, and rock sampling was performed in accordance with the following procedures from the Environmental Procedures Catalog (GJO 6):

- SL–6(P), “Technical Comments on ASTM D 1452–80(90)—Standard Practice for Soil Investigation and Sampling by Auger Borings.”
- SL–7(P), “Technical Comments on ASTM D 1586–84(92)—Standard Test Method for Penetration Test and Split-Barrel Sampling of Soils.”
- SL–9(P), “Technical Comments on ASTM D 2113–83(93)—Standard Practice for Diamond Core Drilling for Site Investigation.”
- GN–8(P), “Standard Practice for Sample Labeling.”
- GN–9(P), “Standard Practice for Chain-of-Sample-Custody Control and Physical Security of Samples.”
- GN–13(P), “Standard Practice for Equipment Decontamination.”

4.6.1 Lithologic Logging

Lithologic logging was performed on site by the field geologist on all core samples that were collected from the unconsolidated materials and bedrock formation during the course of the permanent monitor well installations. Lithologic descriptions of the core samples are presented in the borehole summaries in Appendix A. All lithologic logging was performed in accordance with the following procedures from the Environmental Procedures Catalog (GJO 6):

- SL–19(P), “Technical Comments on ASTM D 2488–93—Standard Practice for Description and Identification of Soils.”
- SL–24(P), “Technical Comments on ASTM D 2487–93—Standard Classification of Soils for Engineering Purposes (Unified Soil Classification System).”

4.7 Distribution Ratio (R_d) Analysis

The distribution coefficient (K_d) is a bulk parameter that has been used with some success to describe the retardation of contamination in an aquifer system. Most numerical ground water models use the K_d concept in simulations of contaminant transport. The K_d is approximated from the empirical distribution ratio (R_d). The R_d is determined from laboratory testing. Generally, it is assumed that K_d is equivalent to R_d. The assumption is valid if (1) the system is always in chemical equilibrium, (2) if adsorption is the only chemical mechanism, and (3) if the R_d (or K_d) is independent of solution composition.

Laboratory measurements to determine the R_d for selected analytes were performed on alluvial and bedrock material to support computer-modeling efforts in characterizing subsurface contaminant transport at the Moab site. The approach used in determining R_d values is to perform batch laboratory tests. The tests consist of reacting a mass of uncontaminated alluvium with water containing the contaminants of interest. The water used in the tests is synthesized to approximate the composition of ground water upgradient of the site. The amount of contaminant that sorbs to the solid matrix (as indicated by decreases in aqueous concentration) is a measure of the tendency for contaminants to be retarded by the aquifer. Details regarding the laboratory procedure used to determine the R_d values are presented in Appendix D, Calculation X0023200. A summary of the results is provided in the following sections.

4.7.1 Method of Solution

Laboratory analyses of the R_d were performed according to ESL procedure CB(Rd–1) (GJO 210), which closely follows American Society for Testing and Materials (ASTM) batch-type method procedure D 4646-87 (ASTM 2001). Essentially, the ESL procedure involves placing a sample representative of a site (e.g., soil, sediments, bedrock) into a solution containing simulated contaminated ground water with which the material is likely to come in contact. The simulated ground water solution is agitated for 24 hours and filtered. The filtered solution is analyzed and compared to the contaminant concentrations of the original solution. The difference between the two is assumed to be adsorbed to the sample. The linear adsorption isotherm distribution coefficient is generally defined as

$$C_{soil} = R_d \times C_{water}, \text{ which can be rearranged to } R_d = C_{soil}/C_{water},$$

or the ratio of the concentration of the contaminant in soil (or other material of interest) to the concentration of the contaminant in water at equilibrium. Therefore, the higher the R_d, the greater the retardation of a contaminant in ground water.

The procedure requires analysis of only the solutions (and no actual soil samples) used in the experiments. Site samples collected from background areas or uncontaminated site samples are generally used, and all contaminant loss in the final solution is attributed to sample adsorption. Use of contaminated samples could potentially underestimate the R_d for contaminants with higher adsorptive properties, because the adsorptive ability of the sample would be reduced by contaminants already present.

4.7.1.1 Sample Selection

Samples for this study were collected from boring MOA-434, located upgradient of the alluvial aquifer plume in an area that is believed to be unaffected by site-related contamination (Figure 4–12). Samples are from depths of 39, 41, 51, and 57 ft, which correspond to 3, 5, 15, and 21 ft below the water table, respectively. The samples are composed of red, silty sand (39 ft), clayey silt (41 ft), red sand (51 ft), and sandy gravel (57 ft).

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Figure 4–12. R_d and Subpile Soil Sample Location Map

4.7.1.2 Sample Preparation

All the samples were air dried at room temperature and sieved to less than 10 mesh (2 millimeters [mm]). The difference between air dried and oven-dried weights is typically less than 1 percent, so no correction was made for the water contents of the air-dried samples.

A synthetic solution was prepared that simulated ground water upgradient of the Moab site, based on ground water analysis for well ATP-3, which was collected November 19, 2000. The synthetic ground water was spiked with uranium (U) and ammonium (NH_4) to simulate interaction with contaminated sediments in the mill tailings area, and the pH was adjusted to about 7.5 (All references regarding concentration data for ammonium are in terms of combined NH_3^0 and NH_4^+ reported as NH_4.) The analysis of the synthetic ground water (which represents the starting composition for the K_d determinations) is as follows (in mg/L): NH_4 (256.1), U (0.966), sodium (325.3), potassium (13.5), calcium (69.9), magnesium (47.8), sulfate (838.4), chloride (503.8), nitrate (19.3), and carbon (inorganic) (42.9); the measured conductivity was about 2,700 microsiemens per centimeter (µS/cm).

Splits of each sample were weighed (1, 2, 5, 10, 15, and 20 g) and placed in 125-mL Nalge bottles with 100 mL of the synthetic ground water. Samples were rotated end-over-end at 8 revolutions per minute (rpm) for 24 hours. They were then centrifuged 20 minutes at 3,500 rpm and filtered through a 0.45 µm filter. Leachate samples were preserved with concentrated acid (nitric for U analysis and sulfuric for NH_4 analysis) to less than pH 2 and submitted to the Analytical Chemistry Laboratory for analysis. Three samples of spiked synthetic ground water solution were also analyzed for control.

4.7.1.3 Sample Analysis and Calculations

The R_d values are calculated by:

$$R_d = \frac{(A-B)V}{(Ms)\,B}$$

where

A = initial concentration of the constituent (mg/L),
B = final concentration of the constituent after contact with soil/sediment (mg/L),
V = volume of solution (mL),
Ms = mass of soil used (g), and
R_d = distribution ratio (mL/g).

For example, the R_d for uranium in sample MOA434-39-10 using the less-than-2-mm fraction is derived as follows:

A = 0.966 mg/L
B = 0.809 mg/L
V = 100 mL
Ms = 10 g

$$R_d = \frac{(0.966-0.809) \times 100}{10 \times 0.809}$$

R_d = 1.94 mL/g

Calculated values for the alluvial aquifer samples were adjusted on the basis of aquifer grain size analysis, which was performed on each sample. The amount of sediment with grain size less than 2 mm ranged from 74.5 to 100 percent. Therefore, calculated R_ds for the alluvial aquifer were adjusted by multiplying by 0.745 to 1.00. This assumes that sediment coarser than 2 mm has less adsorption capacity than the finer clay fraction because of the lower surface area per volume and because surface ionic charge of the coarser fraction is insignificant.

4.7.1.4 Discussion of Results

Raw data and results are shown in Table 4–10 for uranium and in Table 4–11 for ammonium. Sample notation consists of the sample location designation (MOA-434) followed by sample depth and mass of sample used in the procedure. Samples designated with a "B" are synthetic ground water samples that were run through the same procedure as test samples (without any soil); samples designated with a "P" are synthetic ground water samples that were not run through the procedure.

Uranium R_d values, corrected for grain size, are mostly between 1 and 3 mL/g. The values are similar to, perhaps just slightly higher than, values from most UMTRA Project sites. In contrast, the ammonium R_d values are quite varied. R_d values for NH_4 are mostly between 0.5 and 3 mL/g, particularly for the results obtained using the 5 g sample size (with a 1:20 soil to solution ratio, as specified in the ASTM method). These results are similar to results from other studies done on NH_4 at the ESL. In general, significantly lower R_d values are associated with tests using larger amounts of sediment, suggesting that the presence of sediment has some effect on the final dissolved concentrations. Some of this effect could be due to adsorption, but some or most is likely the result of degradation or volatilization of NH_4 during the testing. A literature search was conducted to obtain other K_d values obtained for ammonia. All literature values for K_d were less than 1 (Ceazan et al. 1989; Kipp 1987), indicating that the low end of the site-specific range determined for Moab might be most applicable. A value of 0.5 mL/g was recommended as a result of the literature search (Appendix D, Calculation X0023000).

Upgradient ground water chemistry was used in the tests to simulate future conditions as the ground water system evolves toward a more dilute concentration. The fact that R_ds are probably higher for this dilute water than they would be for conditions currently in ground water beneath the tailings also supports using R_d values at the lower end of the range of results.

4.8 Subpile Soil Analysis

Pore fluids draining from the mill tailings pile move through and interact with the soils below the base of the pile ("subpile soils") before the fluids reach the ground water. Contaminants in the tailings pore fluids may become adsorbed onto the subpile soils or precipitated from solution, or the contaminants may move through the subpile soils in a nonreactive manner, depending on the characteristics of the soils and the nature of the contaminants in the pore fluids. If contaminants become concentrated in the subpile soils, these soils can act as a long-term source of ground water contamination.

Even if the tailings and subpile soils are removed to meet UMTRCA radiological cleanup standards, residual contamination could remain below the depth of remediation at levels that can affect ground water quality. Soil samples were collected and tested to characterize the conditions in the subpile soils and to evaluate the potential for these subpile soils to act as long-term sources of ground water contamination. Details regarding the laboratory procedure used to evaluate the potential for residual contaminants in the subpile soils and a comparison to historical subpile soil data collected by several other organizations, which included ORNL; Steffen, Robertson and Kirsten (SRK); and Shepherd Miller, Inc., are presented in Appendix D, Calculation X0023400). A summary of DOE's analytical results is provided in the following sections.

4.8.1 Subpile Soil Sample Selection

Subpile soil samples were collected during the summer 2002 field season from three borings advanced through the Moab tailings pile and from one background location (pile locations MOA-437, -438, -439; background location MOA-434; Figure 4–12). Lithologic logs for these borings are provided in Appendix A. Samples were collected from several depth intervals at each location.

4.8.2 Subpile Soil Sample Preparation, Extraction, and Analysis

Chemical extractions were used to determine the potential mobility of contaminants. Samples were subjected to a 5 percent acid leach (resulting in a solution pH of <2) according to ESL procedure CB(BT-1) (GJO 210). Decantate samples (i.e., fluids recovered from the subpile soils after the leaching procedure was complete) were submitted to the DOE's Analytical Chemistry Laboratory in Grand Junction for analysis of ammonia, arsenic, iron, manganese, molybdenum, selenium, sulfate, uranium, and vanadium. These constituents were selected because of their prevalence in the tailings pore fluids as determined by previous analyses (SRK 2000). Laboratory results for the current study are provided in Table 4–12.

Analyses for radium-226 were completed on samples from the same boreholes (pile locations MOA-437, -438, and -439 and background location MOA-434) shown on Figure 4–12. These analyses were performed using the Opposed Crystal System (OCS) according to standard procedure RD-4(T) (GJO 6). See the procedure for details, including equations for calculating true concentrations of radium-226. Results of the OCS analysis, sample depths, and soil descriptions are presented in Table 4–13. The same samples were analyzed with two different instruments (MCB1 and MCB2) to cross check; analyses were similar for each.

4.8.3 Calculation

Two grams of soil sample were extracted with a total of 200 mL of acid solution according to ESL Procedure CB(BT-1) (GJO 210). The resulting concentrations in micrograms per liter (µg/L, Table 4–12) of constituents measured in each decantate sample were then converted to units of milligrams per kilograms (mg/kg) to estimate the amount of extractable contaminant per mass of subpile soil by the following equation:

$$\frac{200\text{ mL acid solution}}{2\text{ g sediment}} \times \frac{\mu\text{g contaminant}}{\text{L decantate}} \times \frac{\text{L}}{1{,}000\text{ mL}} \times \frac{\text{mg}}{1{,}000\ \mu\text{g}} \times \frac{1{,}000\text{ g}}{\text{kg}} = \text{mg/kg}$$

Table 4–10. Summary of R_d Data for Uranium

Uranium Sample	V[a] (mL)	Ms[b] (g)	A[c] (mg/L)	B[d] (mg/L)	R_d[e] (mL/g)	Percent of < 2mm	Cor[f] R_d (mL/g)	Unit
MOA434-39-1	100	1	0.966	0.965	0.10	89.80	0.09	Red, silty sand, 3 ft below water
MOA434-39-2	100	2	0.966	0.933	1.77	89.80	1.59	Red, silty sand, 3 ft below water
MOA434-39-5	100	5	0.966	0.881	1.93	89.80	1.73	Red, silty sand, 3 ft below water
MOA434-39-10	100	10	0.966	0.809	1.94	89.80	1.74	Red, silty sand, 3 ft below water
MOA434-39-15	100	15	0.966	0.753	1.89	89.80	1.69	Red, silty sand, 3 ft below water
MOA434-39-20	100	20	0.966	0.716	1.75	89.80	1.57	Red, silty sand, 3 ft below water
MOA434-41-1	100	1	0.966	0.934	3.43	100.00	3.43	Clayey silt, 5 ft below water
MOA434-41-2	100	2	0.966	0.895	3.97	100.00	3.97	Clayey silt, 5 ft below water
MOA434-41-5	100	5	0.966	0.847	2.81	100.00	2.81	Clayey silt, 5 ft below water
MOA434-41-10	100	10	0.966	0.788	2.26	100.00	2.26	Clayey silt, 5 ft below water
MOA434-41-15	100	15	0.966	0.716	2.33	100.00	2.33	Clayey silt, 5 ft below water
MOA434-41-20	100	20	0.966	0.672	2.19	100.00	2.19	Clayey silt, 5 ft below water
MOA434-51-1	100	1	0.966	0.935	3.32	100.00	3.32	Sand, red, 15 ft below water
MOA434-51-2	100	2	0.966	0.918	2.61	100.00	2.61	Sand, red, 15 ft below water
MOA434-51-5	100	5	0.966	0.868	2.26	100.00	2.26	Sand, red, 15 ft below water
MOA434-51-10	100	10	0.966	0.821	1.77	100.00	1.77	Sand, red, 15 ft below water
MOA434-51-15	100	15	0.966	0.764	1.76	100.00	1.76	Sand, red, 15 ft below water
MOA434-51-20	100	20	0.966	0.704	1.86	100.00	1.86	Sand, red, 15 ft below water
MOA434-57-1	100	1	0.966	0.926	4.32	74.50	3.22	Sandy gravel, 21 ft below water
MOA434-57-2	100	2	0.966	0.913	2.90	74.50	2.16	Sandy gravel, 21 ft below water
MOA434-57-5	100	5	0.966	0.878	2.00	74.50	1.49	Sandy gravel, 21 ft below water
MOA434-57-10	100	10	0.966	0.829	1.65	74.50	1.23	Sandy gravel, 21 ft below water
MOA434-57-15	100	15	0.966	0.772	1.68	74.50	1.25	Sandy gravel, 21 ft below water
MOA434-57-20	100	20	0.966	0.729	1.63	74.50	1.21	Sandy gravel, 21 ft below water
MOA434-B1	100	0	na	0.954	na	na	na	na
MOA434-B2	100	0	na	0.946	na	na	na	na
MOA434-P1	100	0	na	0.951	na	na	na	na

[a] V = volume of solution (mL)
[b] Ms = mass of soil used (g)
[c] A = initial concentration of the constituent (mg/L)
[d] B = final concentration of the constituent after contact with soil/sediment (mg/L)
[e] R_d = distribution ratio (mL/g)
[f] Cor = corrected for grain size

Table 4–11. Summary of R_d Data for Ammonium

Ammonium Sample	V[a] (mL)	Ms[b] (g)	A[c] (mg/L)	B[d] (mg/L)	R_d [e] (mL/g)	Percent of < 2mm	Cor[f] R_d (mL/g)	Unit
MOA434-39-1	100	1	256.1	235.0	8.98	89.80	8.06	Red, silty sand, 3 ft below water
MOA434-39-2	100	2	256.1	248.0	1.63	89.80	1.47	Red, silty sand, 3 ft below water
MOA434-39-5	100	5	256.1	239.0	1.43	89.80	1.29	Red, silty sand, 3 ft below water
MOA434-39-10	100	10	256.1	235.0	0.90	89.80	0.81	Red, silty sand, 3 ft below water
MOA434-39-15	100	15	256.1	228.0	0.82	89.80	0.74	Red, silty sand, 3 ft below water
MOA434-39-20	100	20	256.1	219.0	0.85	89.80	0.76	Red, silty sand, 3 ft below water
MOA434-41-1	100	1	256.1	244.0	4.96	100.00	4.96	Clayey silt, 5 ft below water
MOA434-41-2	100	2	256.1	235.0	4.49	100.00	4.49	Clayey silt, 5 ft below water
MOA434-41-5	100	5	256.1	223.0	2.97	100.00	2.97	Clayey silt, 5 ft below water
MOA434-41-10	100	10	256.1	200.0	2.81	100.00	2.81	Clayey silt, 5 ft below water
MOA434-41-15	100	15	256.1	179.0	2.87	100.00	2.87	Clayey silt, 5 ft below water
MOA434-41-20	100	20	256.1	164.0	2.81	100.00	2.81	Clayey silt, 5 ft below water
MOA434-51-1	100	1	256.1	255.0	0.43	100.00	0.43	Sand, red, 15 ft below water
MOA434-51-2	100	2	256.1	245.0	2.27	100.00	2.27	Sand, red, 15 ft below water
MOA434-51-5	100	5	256.1	240.0	1.34	100.00	1.34	Sand, red, 15 ft below water
MOA434-51-10	100	10	256.1	232.0	1.18	100.00	1.18	Sand, red, 15 ft below water
MOA434-51-15	100	15	256.1	219.0	1.13	100.00	1.13	Sand, red, 15 ft below water
MOA434-51-20	100	20	256.1	212.0	1.04	100.00	1.04	Sand, red, 15 ft below water
MOA434-57-1	100	1	256.1	251.0	2.03	74.50	1.51	Sandy gravel, 21 ft below water
MOA434-57-2	100	2	256.1	256.0	0.02	74.50	0.01	Sandy gravel, 21 ft below water
MOA434-57-5	100	5	256.1	248.0	0.65	74.50	0.49	Sandy gravel, 21 ft below water
MOA434-57-10	100	10	256.1	236.0	0.85	74.50	0.63	Sandy gravel, 21 ft below water
MOA434-57-15	100	15	256.1	227.0	0.85	74.50	0.64	Sandy gravel, 21 ft below water
MOA434-57-20	100	20	256.1	226.0	0.67	74.50	0.50	Sandy gravel, 21 ft below water
MOA434-B1	100	0	na	253.0	na	na	na	na
MOA434-B2	100	0	na	249.0	na	na	na	na
MOA434-P1	100	0	na	252.0	na	na	na	na

[a] V = volume of solution (mL)
[b] Ms = mass of soil used (g)
[c] A = initial concentration of the constituent (mg/L)
[d] B = final concentration of the constituent after contact with soil/sediment (mg/L)
[e] R_d = distribution ratio (mL/g)
[f] Cor = corrected for grain size

Table 4–12. Analytical Results for Subpile Soil Decantate[a]

Sample ID[b]	NH_4 (µg/L)	As (µg/L)	Fe (µg/L)	Mn (µg/L)	Mo (µg/L)	Se (µg/L)	SO_4 (µg/L)	U (µg/L)	V (µg/L)	Description
MOA 434-39	34.6	7.1	5,160	2,210	6.6	0.11	2,350	27.2	20.8	Silty sand, red
MOA 434-41	41.5	13.6	16,600	2,820	<3	<0.1	3,100	7.8	37.6	Clayey silt, 5 ft below water
MOA 434-51	17.4	7.2	8,140	1,790	<3	<0.1	770	4.1	32.5	Sand, red, 15 ft below water
MOA 434-57	24.2	8.4	8,680	1,220	<3	<0.1	1,590	4.5	30	Sandy gravel, 21 ft below water
MOA 437-41	2,850	113	48,200	5,380	16.6	1.8	12,600	892	4,330	Tailings, at contact
MOA 437-43	409	19.8	2,650	2,620	8.7	0.56	2,700	69.8	416	Sand/silt, 2 ft below contact
MOA 437-46	86.2	17.2	1,870	3,560	6.1	0.27	2,560	13.1	267	Sand/silt, 5 ft below contact
MOA 437-51	31.6	8.1	1,350	2,250	4.6	0.48	1,910	14.6	151	Sand, 10 ft below contact
MOA 437-82	31.3	23	3,230	2,700	8.4	0.15	3,810	18.5	213	Silt, 50 ft below tailings, 6 ft above water
MOA 437-91	34.6	10.7	2,810	3,670	27.7	0.32	10,900	17.3	50.8	3 ft below water
MOA 437-110	28.6	5.5	6,350	1,310	6.8	0.6	5,740	14.1	13.8	Sandy gravel, sample biased with fines, 54 ft below tailings, 12 ft below water
MOA 438-72	5,620	124	16,500	332	464	2.1	1,430,000	971	2,180	Tailings at contact
MOA 438-75	1,290	8.8	4,590	4,740	13.4	0.49	75,900	34	108	Sand, red, 2 ft below tailings
MOA 438-80	628	3.2	987	3,420	5.4	<0.1	4,810	8.3	30.9	Sand, red 7 ft below tailings
MOA 438-90	556	5.6	7,150	4,680	11.6	<0.1	16,800	13.5	39	Sand, red, 12 ft below tailings, 6 ft above water
MOA 438-101	168	4	1,360	3,420	6.3	<0.1	6,210	6.6	13.4	Sand, red, 23 ft below tailings, 5 ft below water
MOA 438-110	124	2.6	5,710	1,170	<3	1.2	5,300	8.1	35.9	Sandy gravel, sample biased with fines, 14 ft below water
MOA 439-82	2,600	83.5	37,500	258	547	1.1	1,510,000	1,910	12,300	Tailings at contact
MOA 439-83	2,910	8.5	23,700	24,400	<3	0.24	82,200	18.2	36.4	Silty sand, 1 ft below contact
MOA 439-87	2,010	9	10,400	14,400	<3	0.32	93,300	23.8	93.3	Silty sand, 5 ft below contact
MOA 439-90	853	3.5	1,370	8,520	<3	0.1	13,100	9	25.4	Sand, 8 ft below contact
MOA 439-95	1,530	8.7	2,970	2,840	<3	<0.1	10,900	8.3	23.7	Sand, 13 ft below contact
MOA 439-100	796	4.4	1,130	2,640	3.5	0.14	11,200	7	21.5	Sand, 18 ft below contact, 2 ft below water
MOA 439-116	106	8.8	8,510	1,080	3.3	0.47	13,500	7.7	23.5	Sand, gray-brown, 34 ft below contact, 18 ft below water
MOA 439-121	66.9	4.7	9,750	950	5	0.33	12,200	4.3	21.2	Gravel, sample biased with fines, 39 ft below contact, 23 ft below water

[a] Liquid used to leach subpile soils.

[b] Last digits in sample identification indicate depth below ground surface (ft).

Notes: NH_4 = ammonium; As = arsenic; Fe = iron; Mn = manganese; Se = selenium; U = uranium; V = vanadium; µg/L = micrograms per liter.

Table 4–13. Ra-226 OCS Gamma Spectral Analysis of Solids

Sample Location	**Radium-226 Concentration (pCi/g)**			**Depth (ft/bgs)**	**Soils Description**
	Analysis 1	**Analysis 2**	**Average**		
434-39	1.5	0.8	1.2	39–39.25	silty sand, red
434-41	2.4	1.9	2.2	41–41.25	clayey silt
434-51	1.9	0.9	1.4	51–51.25	sand, red
434-57	2.8	2.6	2.7	57–57.25	sandy gravel
437-41	2,094.2	2,295.6	2,194.9	40.75–41	tailings
437-42	3.9	4.0	4.0	42–42.25	sand and silt, red
437-43	11.1	11.3	11.2	43–43.25	sand and silt, red
437-44	130.1	140.8	135.5	44–44.25	sand and silt, red
437-45	17.7	19.5	18.6	45–45.25	sand and silt, red
437-46	4.4	0.5	2.5	46–46.25	sand and silt, red
437-47	1.1	2.6	1.9	47–47.25	sand and silt, red
438-73	1,785.0	1,998.3	1,891.7	72.75–73	tailings
438-74	123.5	145.1	134.3	74–74.25	sand, red
438-75	93.6	91.9	92.8	75–75.25	sand, red
438-76	29.6	32.9	31.3	76–76.25	sand, red
438-78	111.8	124.9	118.4	78–78.25	sand, red
439-82	1,993.9	2,321.0	2,157.5	82–82.25	tailings
439-83	2.7	3.9	3.3	83–83.25	silty sand, red
439-84	3.6	3.9	3.8	84–84.25	silty sand, red
439-87	24.0	23.7	23.9	87–87.25	silty sand, red
439-88	1.7	1.4	1.6	88–88.25	sand, red

Notes: pCi/g = picocuries per gram
ft/bgs = feet below ground surface

For example, a concentration of 7.1 µg/L arsenic was measured in the decantate for soil sample MOA 434-39. Converting this value to milligrams per kilogram using the above equation results in 0.71 mg of arsenic per kilogram of subpile soil that is extractable using the acid solution (the calculation is essentially the same as multiplying laboratory results [µg/L] by 0.1).

Soil concentration results along with depths and sample descriptions are provided in Table 4–14. Plots of concentration versus depth are shown on Figure 4–13 through Figure 4–21.

4.8.4 Discussion of Subpile Soil Results

Chemical extractions were used to determine the potential mobility of contaminants. Samples were subjected to a 5 percent acid leach (resulting in a solution pH <2). It is apparent from examination of the plots presented in Figure 4–13 through Figure 4–21 and data summarized in Table 4–14, that most of the analyte concentrations are elevated above background in the tailings and uppermost subpile samples. An exception to this is manganese, which is actually lower than background at locations MOA-438 and -439, but increases to background levels with increasing depth. For all constituents except ammonium, selenium, and sulfate, concentrations decrease rapidly with increasing depth to concentrations within the range of natural background. The ground water elevation at location MOA-437 is approximately 47 ft below the tailings contact, whereas at locations MOA-438 and -439 the elevation is at 23 ft and 16 ft below the contact, respectively. This may explain why ammonium attenuates more rapidly with depth at location MOA-437 compared to MOA-438 and-439. Interaction with ground water

Table 4–14. Converted 2002 Subpile Soil Study Data

Sample ID	NH_4 (mg/kg)	As (mg/kg)	Fe (mg/kg)	Mn (mg/kg)	Mo (mg/kg)	Se (mg/kg)	SO_4 (mg/kg)	U (mg/kg)	V (mg/kg)	Depth (ft)	Dist. From Cont. (ft)	Description
MOA 434-39	3.46	0.71	516	221	0.66	0.011	235	2.72	2.08	39	-60	Silty sand, red
MOA 434-41	4.15	1.36	1,660	282	0.3	<0.01	310	0.78	3.76	41	-65	Clayey silt, 5-ft below water
MOA 434-51	1.74	0.72	814	179	0.3	<0.01	77	0.41	3.25	51	-70	Sand, red, 15 ft below water
MOA 434-57	2.42	0.84	868	122	0.3	<0.01	159	0.45	3	57	-75	Sandy gravel, 21 ft below water
MOA 437-41	285	11.3	4,820	538	1.66	0.18	1,260	89.2	433	41	0	Tailings, at contact
MOA 437-43	40.9	1.98	265	262	0.87	0.056	270	6.98	41.6	43	-2	Sand/Silt, 2 ft below contact
MOA 437-46	8.62	1.72	187	356	0.61	0.027	256	1.31	26.7	46	-5	Sand/Silt, 5 ft below contact
MOA 437-51	3.16	0.81	135	225	0.46	0.048	191	1.46	15.1	51	-10	Sand, 10 ft below contact
MOA 437-82	3.13	2.3	323	270	0.84	0.015	381	1.85	21.3	82	-41	Silt, 50 ft below tailings, 6 ft above water
MOA 437-91	3.46	1.07	281	367	2.77	0.032	1,090	1.73	5.08	91	-50	3 ft below water
MOA 437-110	2.86	0.55	635	131	0.68	0.06	574	1.41	1.38	110	-69	Sandy gravel, sample biased with fines, 54 ft below tailings, 12 ft below water
MOA 438-72	562	12.4	1,650	33.2	46.4	0.21	143,000	97.1	218	72	0	Tailings at contact
MOA 438-75	129	0.88	459	474	1.34	0.049	7,590	3.4	10.8	75	-3	Sand, red, 2 ft below tailings
MOA 438-80	62.8	0.32	98.7	342	0.54	<0.01	481	0.83	3.09	80	-8	Sand, red 7 ft below tailings
MOA 438-90	55.6	0.56	715	468	1.16	<0.01	1,680	1.35	3.9	90	-18	Sand, red, 12 ft below tailings, 6 ft above water
MOA 438-101	16.8	0.4	136	342	0.63	<0.01	621	0.66	1.34	101	-29	Sand, red, 23 ft below tailings, 5 ft below water
MOA 438-110	12.4	0.26	571	117	<0.3	0.12	530	0.81	3.59	110	-38	Sandy gravel, sample biased with fines, 14 ft below water
MOA 439-82	260	8.35	3,750	25.8	54.7	0.11	151,000	191	1,230	82	0	Tailings at contact
MOA 439-83	291	0.85	2,370	2,440	<0.3	0.024	8,220	1.82	3.64	83	-1	Silty sand, 1 ft below contact
MOA 439-87	201	0.9	1,040	1,440	<0.3	0.032	9,330	2.38	9.33	87	-5	Silty sand, 5 ft below contact
MOA 439-90	85.3	0.35	137	852	<0.3	0.01	1,310	0.9	2.54	90	-8	Sand, 8 ft below contact
MOA 439-95	153	0.87	297	284	<0.3	<0.01	1,090	0.83	2.37	95	-13	Sand, 13 ft below contact
MOA 439-100	79.6	0.44	113	264	0.35	0.014	1,120	0.7	2.15	100	-18	Sand, 18 ft below contact, 2 ft below water
MOA 439-116	10.6	0.88	851	108	0.33	0.047	1,350	0.77	2.35	116	-34	Sand, gy-br, 34 ft below contact, 18 ft below water
MOA 439-121	6.69	0.47	975	95	0.5	0.033	1,220	0.43	2.12	121	-39	Gravel, sample biased with fines, 39 ft below contact, 23 ft below water

Notes: NH_4 = ammonium; As = arsenic; Fe = iron; Mn = manganese; Mo = molybdenum; Se = selenium; SO_4 = sulfate; U = uranium; V = vanadium; µg/L = micrograms per liter; mg/kg = milligrams per kilogram; ft = feet at these latter two locations could mean that there is a continued flux of ammonium entering the system or a continual adsorption/desorption process occurring. Alternatively, ammonium may attenuate more rapidly with depth at MOA-437 because of biodegradation occurring in the vadose zone. Ammonium would have more time to degrade at MOA-437 because there is more vadose zone for pore fluids to travel through until ground water is reached. Also, MOA-437 is upgradient; which could also explain why relatively little ammonium in the shallow ground water at MOA-437. However, although concentrations in subpile soils at locations MOA-438 and -439 do not decrease to background, the concentrations do decrease to very low levels. The same is true for sulfate in subpile soil samples collected at locations MOA-437, -438, and -439.

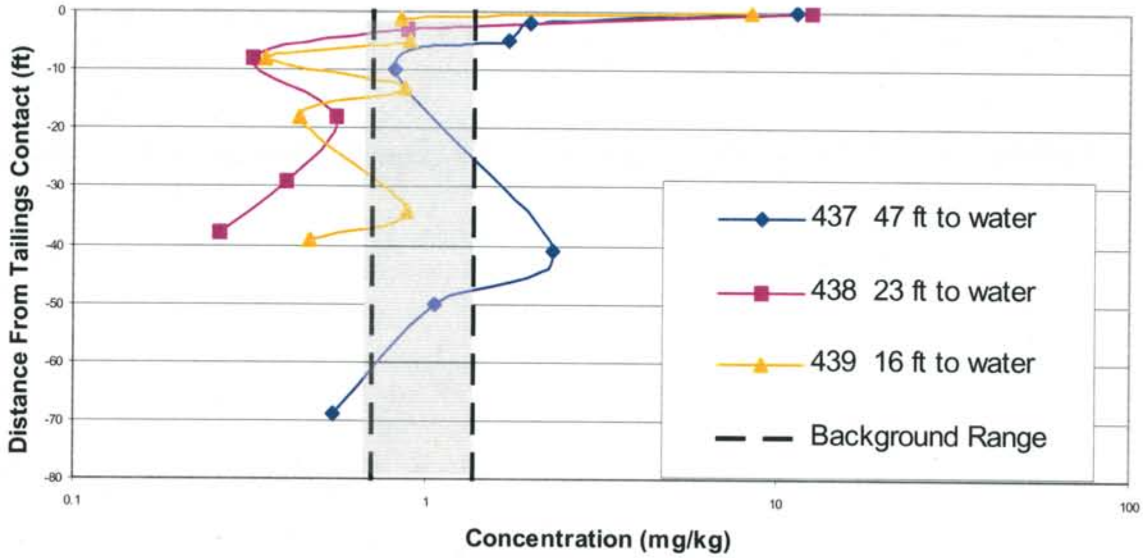

Figure 4–13. Arsenic Concentration Levels

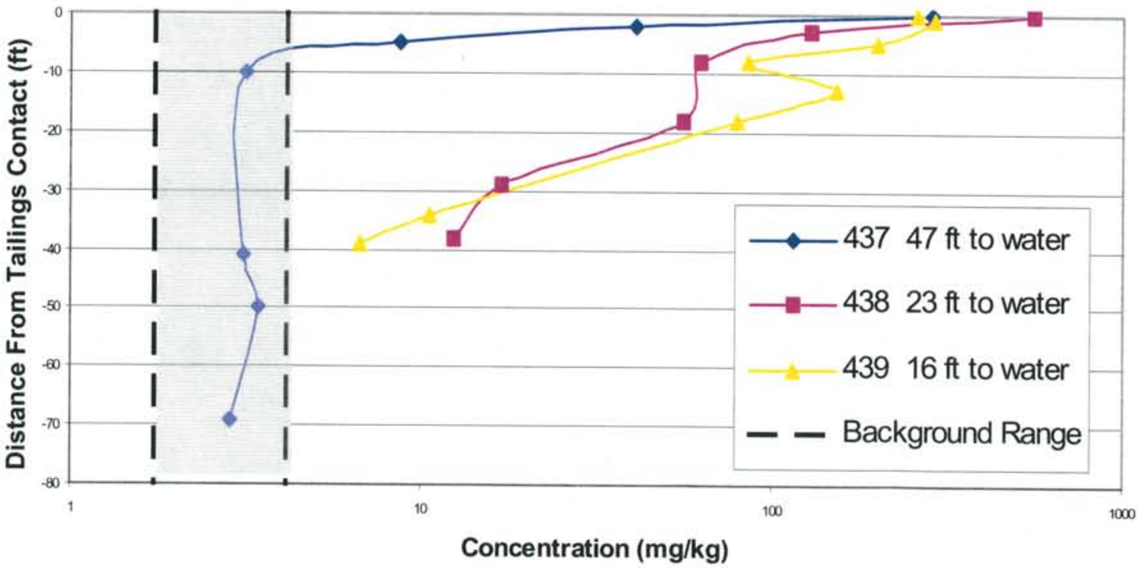

Figure 4–14. Ammonium Concentration Levels

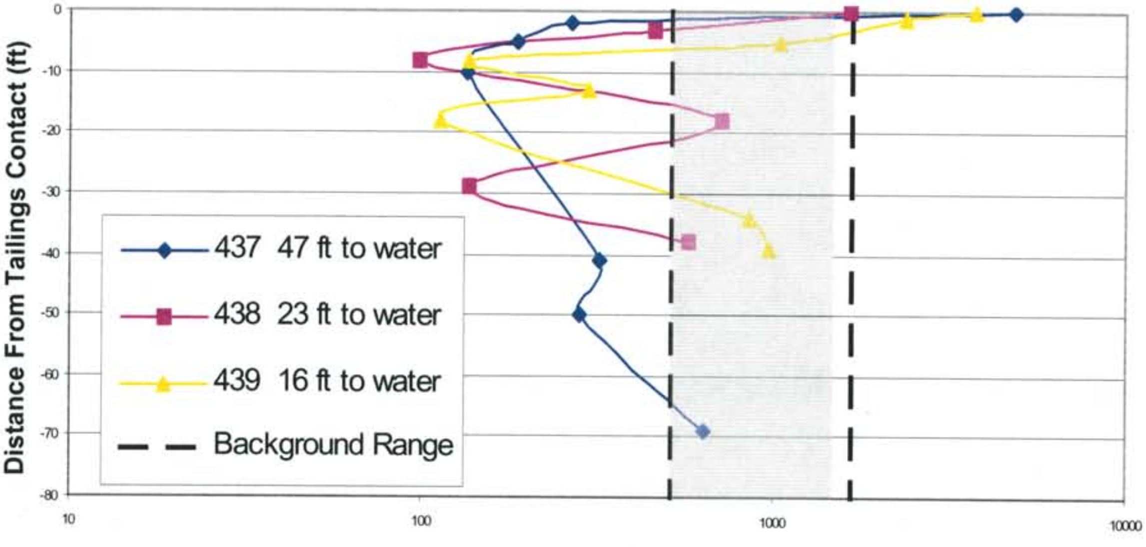

Figure 4–15. Iron Concentration Levels

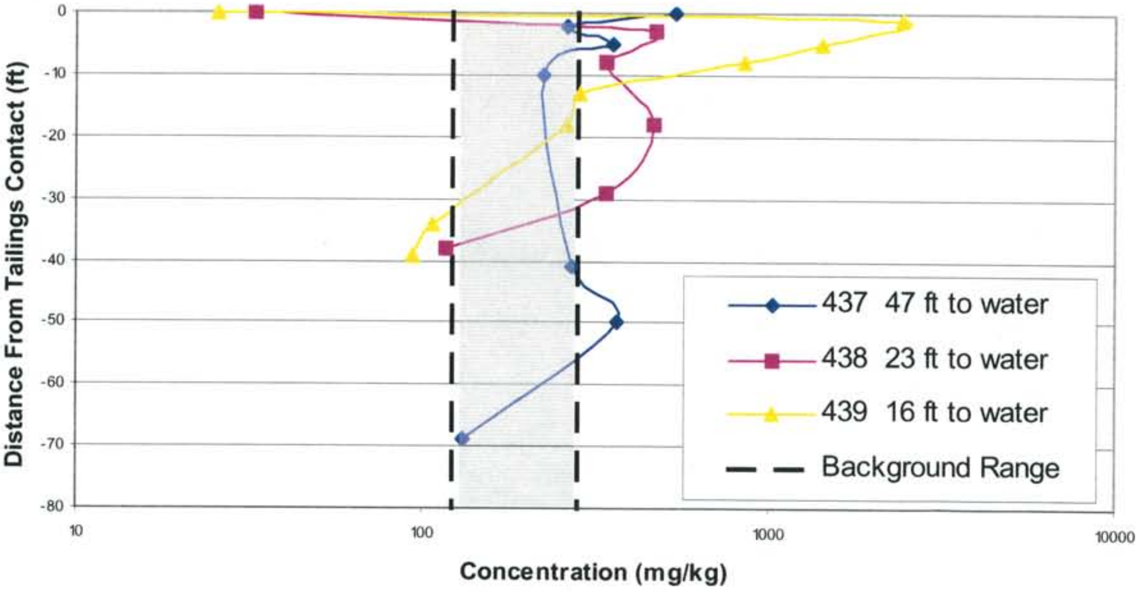

Figure 4–16. Manganese Concentration Levels

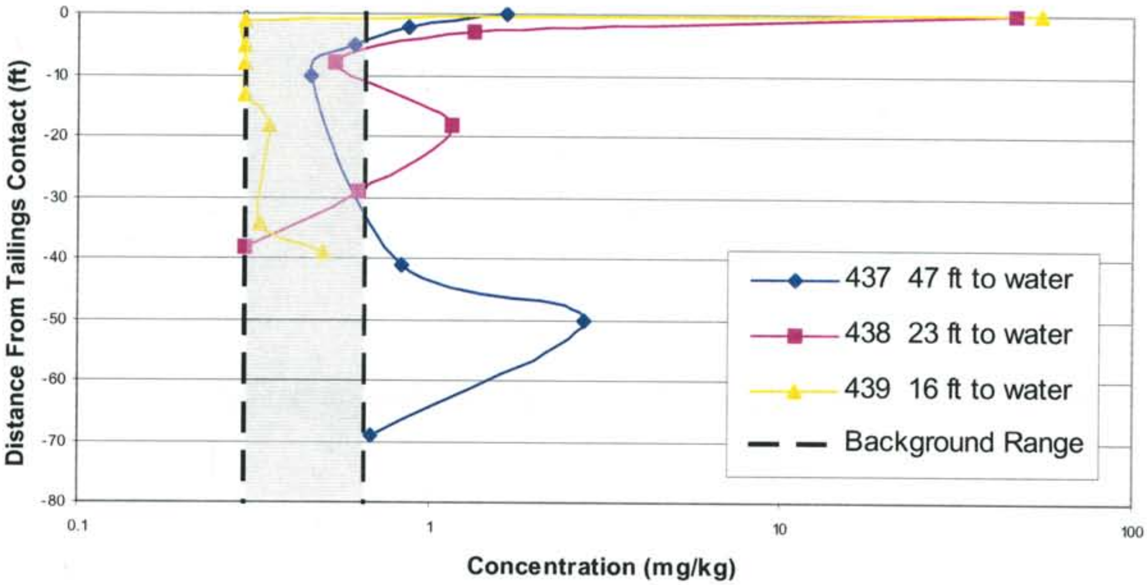

Figure 4–17. Molybdenum Concentration Levels

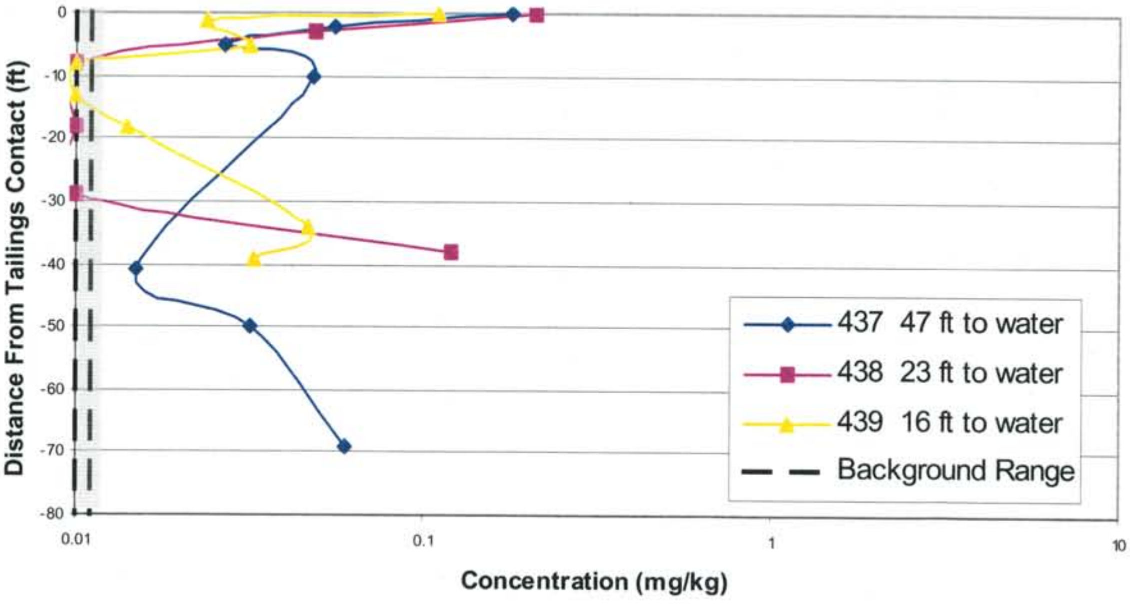

Figure 4–18. Selenium Concentration Levels

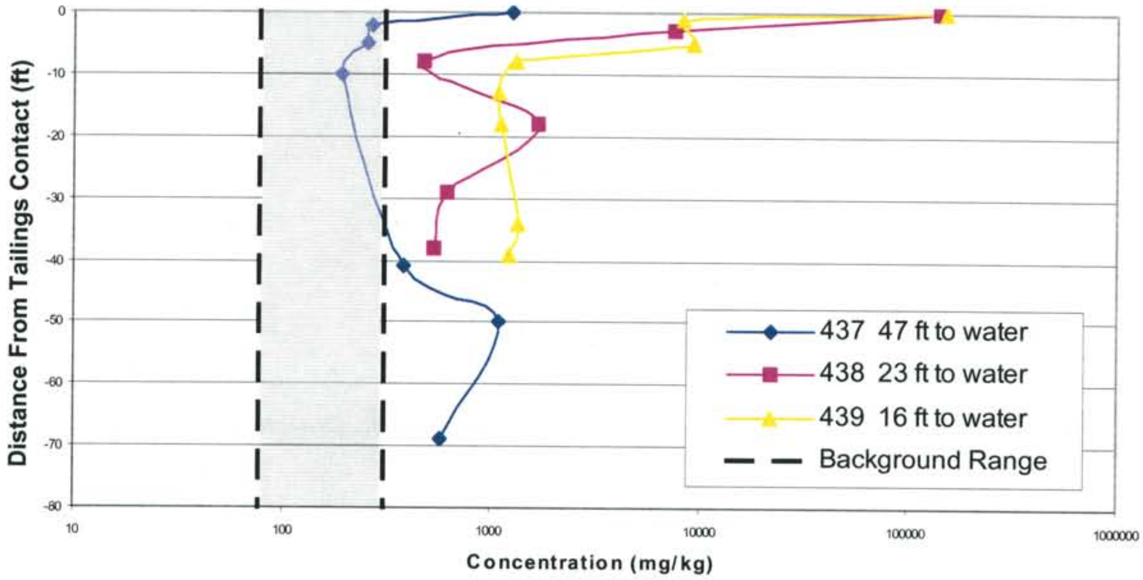

Figure 4–19. Sulfate Concentration Levels

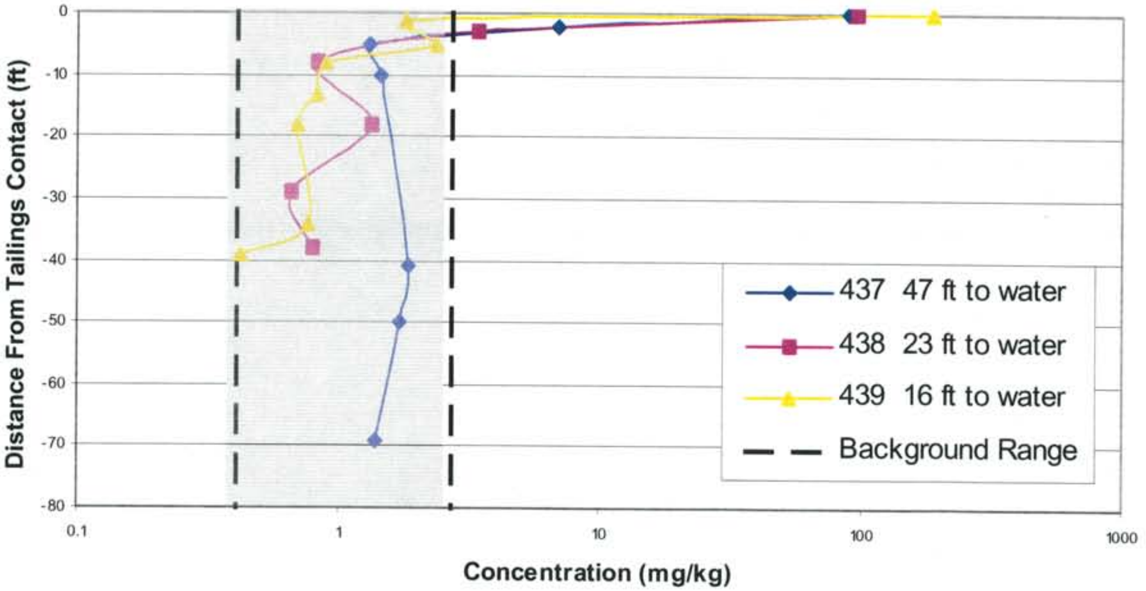

Figure 4–20. Uranium Concentration Levels

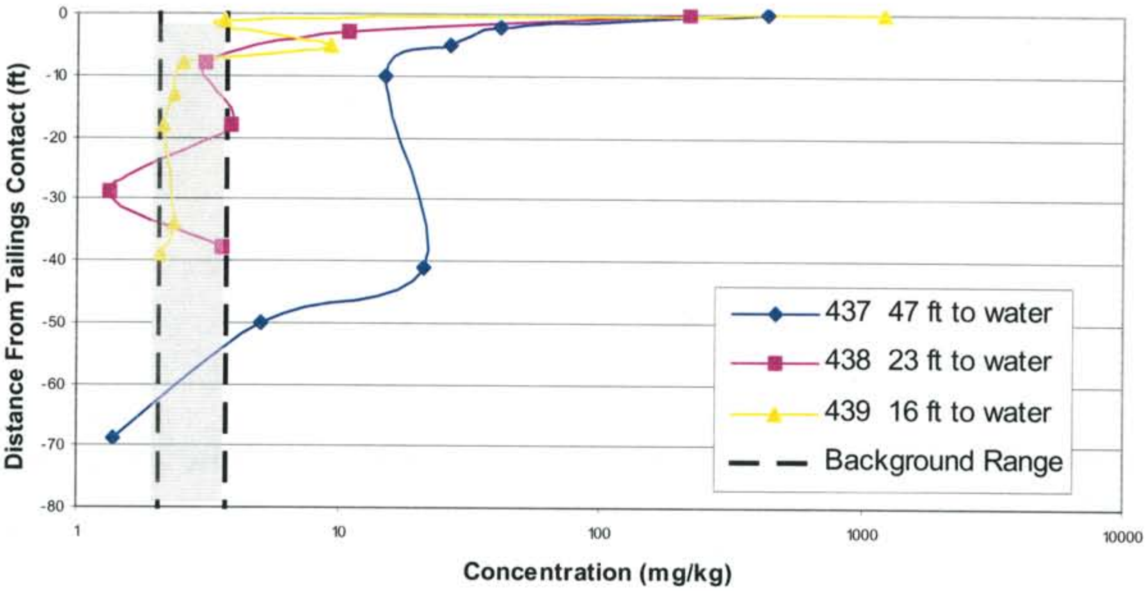

Figure 4–21. Vanadium Concentration Levels

Concentration of several constituents, including iron, fluctuate significantly with changing depth. Because this behavior is seen with iron, it suggests that some type of oxidation-reduction reactions may be responsible, although it could be simple heterogeneity. However, it would be difficult to predict exactly what mechanism may be at work based on solids data only. Only very limited ground water data are available for the depths from which subpile soil samples were collected.

The radium-226 data show a similar, though even more dramatic decrease with depth at most locations (Table 4–13). At locations MOA-437 and -439, radium concentrations decrease to levels below the surface standard of 5 pCi/g within 6 ft of the base of the tailings. (Even though the EPA standard in 40 CFR 192 is actually 5 pCi/g above background, OCS measurements are compared to the 5 pCi/g standard without background to account for measurement uncertainty.) Concentrations of radium-226 at location MOA-438 also decrease within this depth range, though not to levels below the standard. At this location, concentrations remain above the EPA soil cleanup standard to depths of at least 5 ft below the tailings contact.

Common concentration ranges of selected constituents in Moab subpile soils are presented in Table 4–15 for comparison with ranges in Moab site samples. Some constituents such as uranium are consistently higher in Moab subpile samples than naturally occurring concentrations and probably site-related. Other more common constituents such as iron and manganese are well within common ranges for soil and may or may not be attributable to processes at the site. Also provided in Table 4–15 are EPA's Region III soil screening levels for protection of ground water (EPA 2002). Two values are provided—one with a dilution attenuation factor (DAF) of 1, the other with a DAF of 20. These concentrations represent soil concentrations that would result in acceptable ground water concentrations with those respective DAFs. A DAF of 1 assumes no attenuation of contaminants occurs as ground water moves through the subsurface; this assumption is not realistic. EPA has selected a DAF of 20 as the "default" DAF to account for the variety of processes that affect contaminant movement in the subsurface. The default soil screening levels for ground water also assume a 0.5 acre source area. Site-specific data, if available, can be used to develop site-specific and contaminant-specific DAFs. When default DAFs are used for comparison, only arsenic exceeds its DAF, which is very low due to arsenic's high toxicity.

Table 4–15. Common Concentration Ranges of Selected Constituents in Soils and Soil Screening Levels

Constituent	Range for Moab Subpile Soils (mg/kg)	Common Range in Soils (mg/kg)	Soil Screening Level for Ground water; 1 DAF[c] (mg/kg)	Soil Screening Level for Ground water; 20 DAF[c] (mg/kg)
Arsenic	0.32–2.3	1−50[a]	0.0013	0.026
Barium	na	100−3,000[b]	110	2,100
Copper	na	2−100[a]	530	11,000
Iron	98.7–2,370	7,000−550,000[a]	na	na
Manganese	95–2,440	20−3,000[a]	330	6,700
Molybdenum	<0.3–2.77	1−5[b]	na	na
Selenium	<0.01–0.056	0.1−2[a]	0.95	19
Uranium	0.43–6.98	1 (avg.)[b]	na	na
Vanadium	1.34–41.6	20−500[b]	260	5,100

[a] McLean and Bledsoe 1992
[b] Levinson 1980
[c] DAF = dilution attenuation factor (EPA 2002)
na = not available

4.9 Hydrogeologic Investigations

Hydrogeologic investigations were conducted to accomplish two main goals: (1) characterize the extent of brine upconing in response to pumping a remediation extraction well and (2) provide estimates of aquifer parameters associated with the alluvial aquifer underlying the site. All aquifer tests were conducted at the SMI-PW01 well cluster (Figure 4–22). At this location the lithology changes from an upper fine-grained (silty sand) unit to a coarser-grained (gravelly sand) unit at a depth of approximately 13 ft below ground surface (bgs) (see Appendix A for boring log).

Three separate field investigations were conducted in 2002 to accomplish the goals of the hydrogeologic characterization. The first investigation was conducted to characterize the potential for upconing into the overlying freshwater system when pumping a remediation well. Details of the procedures and results are provided in *Characterization of Groundwater Brine Zones at the Moab Site (Phase 1)* (DOE 2002a). The second and third field efforts were conducted to determine the aquifer transmissivity, hydraulic conductivity, and specific storage parameters. Aquifer test procedures and results conducted to estimate the aquifer parameters are provided in *Aquifer Test Data Analyses (Phase II, Part I)*, October 2002 (Appendix D, Calculation X0021700) and *Aquifer Test Data Analyses (Phase II, Part 2)*, January 2003 (Appendix D, Calculation X0021700). Major findings from the three hydrogeologic field investigations are summarized in the following sections.

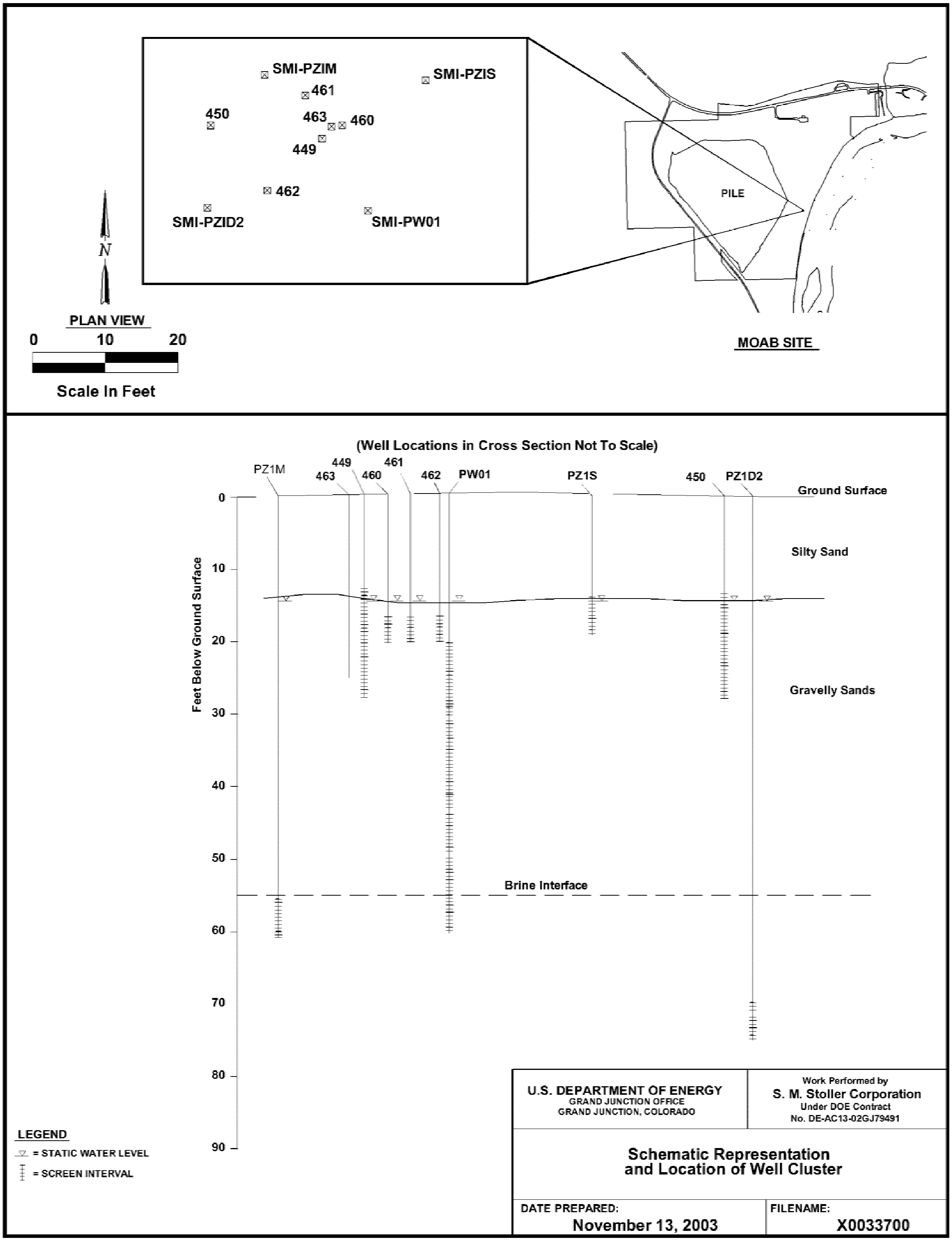

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Figure 4–22. Schematic Representation and Location of Well Cluster

4.9.1 Brine-Upconing Tests

Freshwater in the unconfined alluvial system at the Moab site is underlain by a brine zone. Pumping from the shallow fresher water system (during pump-and-treat remediation) may cause the salt water to rise to a higher elevation and intrude the fresher water. Salt-water intrusion would result in degradation of the overlying fresher water, which could adversely affect the tamarisk plant communities that are providing beneficial phytoremediation at the site (see Section 4.10.3). Besides causing saltwater intrusion into the shallow ground water, rising salt water may bring higher ammonia concentrations to the surface and cause added contamination to the river.

The nature and depth of the brine-freshwater interface at the SMI-PW01 well was characterized so that instrumentation could be properly placed in the pumping and observation wells to measure changes in the location and chemistry of the interface during the test. This was accomplished by collecting baseline ground water samples from the screened interval of the SMI-PW01 well, which is screened from approximately 20 to 60 ft bgs. Ground water grab samples were collected at approximately 5 ft intervals and analyzed for density, TDS, specific conductance, ammonia, chloride, sulfate, and uranium to obtain a vertical concentration profile. Results of the chemical analyses are presented in *Characterization of Groundwater Brine Zones at the Moab Site (Phase 1)* (DOE 2002a).

The interface was further characterized by electrical conductivity measurements collected at location 358, which is adjacent to SMI-PW01 (see Section 4.1). Results of the electrical conductivity profile and the TDS analyses on the ground water grab samples collected at SMI-PW01 are presented in Table 4–2 and plotted in Figure 4–8. Results suggest a sharp interface occurs at approximately 55 ft bgs. At depths less than 55 ft bgs, TDS concentrations range between 10,000 and 20,000 mg/L. At depths between 60 and 80 ft bgs, the TDS concentrations range from 40,000 to 80,000 mg/L.

Once the brine characterization was complete, all tests associated with the brine upconing investigation were conducted at the SMI-PW01 well cluster location. A schematic representation in cross-section and location of the test wells within the cluster is presented in Figure 4–22. Ground water grab samples were collected from each observation well and the pumping wells prior to each test, during the test's pumping periods, and during recovery periods. These samples were also analyzed for density, TDS, specific conductance, ammonia, chloride, sulfate, and uranium. In addition, field parameters (temperature, pH, and specific conductance) were measured on the pumped well discharge water and inside the observation wells using down-hole probes. Two pumping wells were used to conduct the upconing tests. The first series of tests used SMI-PW01 as the pumping well. The second series of tests was conducted using MOA-449 as the pumping well. Results are described below.

4.9.1.1 Pumping Well SMI-PW01

Ground water was pumped from well SMI-PW01 at various flow rates using a submersible pump. These tests were also conducted with the pump intake set at various depths within the screened interval. The first test was conducted at flow rates of 5 and 15 gpm with the pump intake set 23 ft bgs. A second test was conducted at a flow rate of 55 gpm with the intake set 22 ft bgs, and the third test was conducted at a flow rate of 15 gpm with the intake set at 51 ft bgs.

The data collected from these tests indicate that regardless of the flow rate or location of the pump intake, the specific conductance of the discharge water increases when pumping from well SMI-PW01. This increase is probably the result of brine entering the well when the screen extends below the saltwater interface (Figure 4–22), rather than from upconing. These results suggested that a pumping well that was not screened at or below the interface was necessary to evaluate the extent of upconing in response to pumping. For this reason, pumping well MOA-449 was installed.

4.9.1.2 Pumping Well MOA-449

Two long-term aquifer tests were attempted at MOA-449 in August and September 2002 to determine if pumping from a well not screened across or below the saltwater interface will cause upconing of the underlying brine. Pumping well MOA-449 was installed with a screened interval from 13 to 28 ft bgs. At this depth, the bottom of the screen is approximately 27 ft above the top of the saltwater interface. The August 2002 test had to be abandoned due to equipment failure. However, the September 2002 test ran for 17 days at a flow rate of 3 gpm. In the September test, the specific conductance of the discharge water did not increase during the pumping period. This response suggests that water withdrawn from well MOA-449 was originating from the upper portion of the aquifer, and pumping was not affecting the underlying brine.

Short-term aquifer tests were also conducted at higher flow rates (from 4.3 to 7.5 gpm) in November 2002 using well MOA-449 as the pumping well. Ground water parameter data were also collected during these tests to determine if higher flow rates would result in brine upconing. The November 2002 short-term test data also indicated the underlying saltwater interface was not being affected by the higher flow rates from well MOA-449. Even at a pumping rate of 7.5 gpm, the specific conductance of the discharge water did not significantly increase. This suggests that careful extraction well construction, pump depth in the well, and low flow rates, can minimize the effects of brine upconing.

4.9.2 Aquifer Tests

Numerous long-term and short-term aquifer tests were conducted in the vicinity of the SMI-PW01 well cluster (Appendix D, Calculation X0023700). Although, equipment and well efficiency problems affected the results of many of these tests, aquifer tests completed during the November 2002 field effort provided representative aquifer parameter estimates of the upper alluvial aquifer.

Aquifer tests conducted during November 2002 used well 449 as the pumping well. Observation wells 460, 461, and 462 were installed specifically for this investigation (Appendix D, Calculation X0021700). Well 449, which is screened exclusively within the gravelly unit of the aquifer, was installed within the SMI-PW01 well cluster. Three separate, short-term (less than 24 hours) aquifer tests were conducted using three different pumping rates (4.3 gpm, 7.5 gpm, and 7.1 gpm). Drawdown data were analyzed using the Hantush and Jacob Method for Leaky Aquifers and the Distance Drawdown Method. Residual drawdown data collected during recovery tests were analyzed using the Theis Recovery Method.

4.9.2.1 Aquifer Parameter Estimates

Aquifer parameter estimates based on the results of the three short-term aquifer tests are presented in Table 4–16. A more complete discussion is presented in *Aquifer Test Data Analyses (Phase II, Part 2)* (Appendix D, Calculation X0023700).

Table 4–16. Aquifer Test Results from the Gravelly Unit

Test Flow Rate	Well(s)	Analysis Method							
		Hantush and Jacob			Theis Recovery		Distance-Drawdown		
		T (ft^2/day)	K (ft/day)	S	T (ft^2/day)	K (ft/day)	T (ft^2/day)	K (ft/day)	S
4.3 gpm	460	410.2	27.3	0.011	402.0	26.8	na	na	na
	461	810.5	54.0	0.020	1098.2	73.2	na	na	na
7.5 gpm	460	406.0	27.1	0.010	na	na	na	na	na
	461	1023.2	68.2	0.020	na	na	na	na	na
	460/461/462	na	na	na	na	na	399.5	26.6	0.006
7.1 gpm	460	525.7	35.0	0.017	454.4	30.3	na	na	na
	461	1064.4	71.0	0.031	1219.5	81.3	na	na	na
	460/461/462	na	na	na	na	na	520.8	34.7	0.015

Notes: T = Transmissivity; K = Hydraulic conductivity; S = Specific storage; All K values calculated from T, using a saturated thickness of 15 ft
na = not applicable

Analyses of the drawdown data indicate that the transmissivity of the upper basin fill aquifer ranges from 399.5 to 1064.4 ft^2/day. Applying a saturated thickness of 15 ft produces corresponding hydraulic conductivities that range from 26.6 to 71.0 ft/day. Analyses of the drawdown data also indicate that the specific storage ranges from 0.006 to 0.031.

Analyses of residual drawdown data indicate that the transmissivity ranges from 402.0 to 1,219.5 ft^2/day. Again applying a saturated thickness of 15 ft produces the corresponding hydraulic conductivities that range from 26.8 to 81.3 ft/day.

One important finding in addition to the aquifer parameter estimate pertains to the radius of influence measured during the pumping phase of the aquifer tests. The limit of measurable drawdown did not exceed 10.4 ft from the pumping well.

4.10 Ecological Investigations

Field investigations were conducted to evaluate the existing role of tamarisk and other riparian plant species on site hydrology and plume migration and to provide detailed characterization of the sandbar area adjacent to the Moab site. Characterization activities in the sandbar area focused on identifying potentially suitable habitat areas and determining ammonia concentrations discharging from the ground water that may affect sensitive fish species. The latest characterization is more detailed than the studies conducted by Shepherd Miller, Inc. (SMI 2001) and Oak Ridge National Laboratory (ORNL 1998a). Vegetation mapping was conducted to determine plant species distribution and abundance along the floodplain between the toe of the pile and the sandbar area. Plant stresses in response to salinity, ammonia, and ground water fluctuations were measured and mapped. Estimated evapotranspiration rates and spatial patterns were determined.

4.10.1 Potential Suitable Fish Habitat Mapping

Surface and topographic features in the sandbar area were surveyed and a base map constructed for identifying potentially suitable fish habitat areas that may develop into calm backwaters adjacent to the site. The base map was prepared by extracting major features, such as the location of roads, fences, Moab Wash, the Colorado River, and sand bars from a September 2001 aerial photograph of the Moab site. Field mapping using a field grade portable global positioning system (Trimble Model PRO-XRS receiver and TSC-1 data logger) was then performed to provide more detail in the sand bar area. The unit was used with differential correction to achieve an accuracy of about 3 ft. The field mapping was conducted on November 14 and 15, 2001, when the river was at a relatively low flow of 3,190 cfs and 3,160 cfs, respectively (daily means from USGS station 09180500 near Cisco, Utah). Details of the mapping are provided in the characterization report, *Chemistry of Ground Water in the Colorado River Sandbar Area*, (DOE 2002b).

Mapped features are presented in Figure 4–23. For ease of presentation, some distinctive features were given formal names (Cottonwood Island, Cottonwood Island Mud Flat, Moab Wash Delta, North Shallows, South Shallows, and Boulder Flats). Other features were mapped informally using lithologic descriptors (mud ridges, gravel, and tamarisk ridges). The surface features are ephemeral and are modified each spring by runoff in the Colorado River. On the basis of observations of historical aerial photographs, however, it appears that many of the surface features have been similar since major flooding in 1994. The 1994 flooding caused major erosion and redeposition of the sandbar area. The sandbar area is bordered on the west by a 10-ft-high bank that rises steeply up to Bank Road and on the east by the Colorado River.

Two ridges with heavy tamarisk growth are located along the river bank between Moab Wash and the North Shallows. The tamarisk growth provides protection from erosion, and as a result has formed ridges that are elevated approximately 2 to 4 ft above the low-lying flat areas in the river bed to the east and a dry channel to the west. A site visit with the Utah Division of Wildlife Resources was conducted on December 19, 2001, to evaluate and map the most likely areas adjacent to the site that could provide suitable fish habitat for endangered young-of-the-year and adult Colorado pikeminnow and razorback sucker (DOE 2002i). Three areas were identified along the channel that have potential to develop into suitable habitat during a normal river runoff based on defined physical characteristics of: (1) backwater or slow-moving eddies, (2) sandy-silt substrate, and (3) water depths of less than 2-ft (USF&WS 1998). However, due to the seasonally low river stage, little, if any, suitable habitat was present during the site visit. The potentially suitable areas extend approximately 700-ft between Moab Wash and the North Shallows and are delineated in Figure 4−23 as areas A, B, and C. These three areas coincide closely with three areas (4A, 4B, and 4C) that were identified in 1995 during a multi-agency site visit to develop objectives and protocols for a proposed plan to study the effects on the former mill site operations on the Colorado River habitat (DOI 1995). The locations adjacent to the Moab site that were identified in 1995 are presented in a map prepared by WestWater Engineering (1995). DOI characterized these three areas adjacent to the site as “poor endangered species habitat” (DOI 1995), though it noted that fish would need to navigate through waters in those areas.

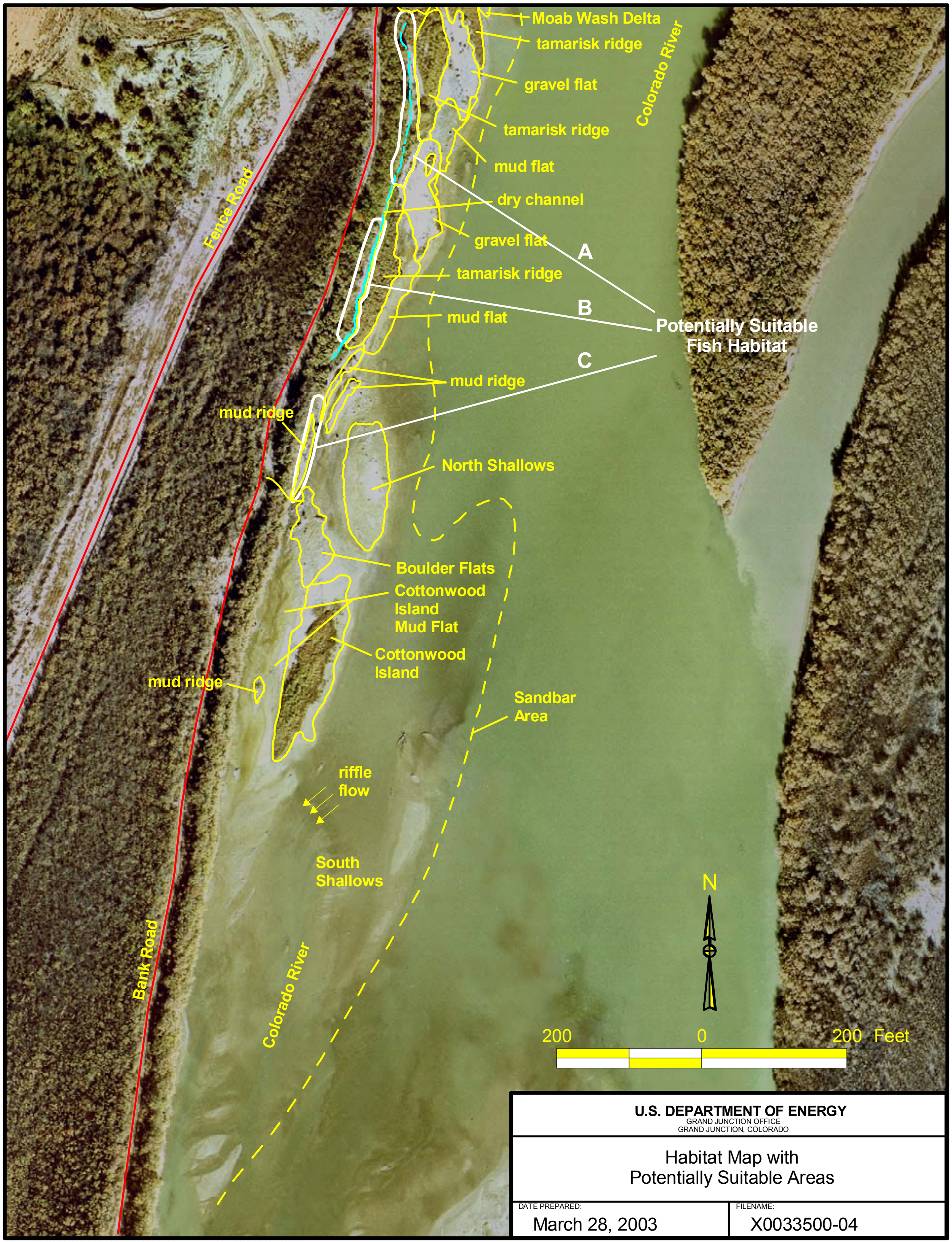

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Figure 4–23. Potentially Suitable Habitat Areas in the Sandbar

4.10.2 Ammonia Distribution Mapping

Characterization of the ammonia distribution in the sandbar area was conducted in December 2001 and August 2002. Field activities during the December 2001 event focus mostly on shallow ground water sampling (pit sampling) of the area where the alluvial aquifer discharges to potentially suitable habitat areas. Samples were collected for laboratory analyses of other constituents including uranium, chloride, and sulfate, and field measurements of pH, oxidation-reduction potential (ORP), temperature, specific conductance, and dissolved oxygen were performed. Details of the December 2001 sampling and analysis are provided in the characterization report *Chemistry of Ground Water in the Colorado River Sandbar Area* (DOE 2002b). Analytical results of the shallow ground water samples are summarized in Appendix C.

The August 2002 sampling was less extensive than the December 2001 sampling and focused on sampling and analyses of river water collected near the shore where elevated ammonia concentration in the shallow ground water mixes with the river water. The sampling was performed August 6 and 7, 2002, when the river flow was relatively low at 1,770 cfs and 2,230 cfs, respectively (daily means from USGS station 09180500 near Cisco, Utah). Field parameters (pH, ORP, temperature, and specific conductance) and ammonia analyses were performed. For comparison, some of the shallow ground water sample locations established in December 2001 were also sampled in August 2002. Results of the ammonia sampling in the sandbar area performed in August 2002 are summarized in Appendix C and in the following sections.

4.10.2.1 Sampling and Analysis Methods

Ground water was sampled from nine shallow test pits previously sampled in December 2001. Prior to the August 2002 sampling, each pit was cleaned out with a shovel until the hole was 1.5 to 3.5 ft in depth. Water flowed into the open holes and filled up to at least 0.5 ft in depth, usually in a few minutes. Field measurements of ammonia, pH, ORP, temperature, and specific conductance were performed within 30 minutes after digging a hole. Instrumentation consisted of a YSI Model 650 attached to a YSI 6290 Sonde equipped with a YSI 6883 NH4 ion-selective electrode. Measurements were performed using ESL procedures AP(pH-1), AP(ORP-1), and AP(EC-1); (GJO 210). The probes were swirled gently in the water in the open pit until the readings stabilized. A short length of 4-inch diameter PVC pipe was placed in some holes to prevent the walls from collapsing on the probes. Specific conductance and pH readings were compensated for temperature.

Immediately after the field measurements were performed, a 500-mL sample was collected from the shallow test pit by pumping ground water through a 0.45-µm filter using a peristaltic pump. Unfiltered samples were also collected. Samples were kept on ice (no other preservation) and analyzed at the ESL for ammonia as N within 24 hours of collection. Ammonia was analyzed by the aminosalicylate colorimetric method (ESL procedures AP[NH3-1] and AP[NH3-2]) (GJO 210).

Field measurements of ammonia, pH, conductivity, and temperature in the river water were performed at 11 locations. The YSI probe was lowered into the water about 3 to 4 inches from the bottom and analyzed in the same manner as the ground water samples. Samples of river water were collected at the same 11 locations and analyzed at the ESL in the same manner as the ground water samples.

The YSI 6883 ion-selective electrode, used for the field analysis of ammonia, measures only NH_4. The meter has internal algorithms that calculate NH_3 concentration using the NH_4 concentration, pH, and temperature. Total ammonia concentration is determined as the sum of the NH_4-N and NH_3-N concentrations. A limited amount of controlled testing was performed at the ESL before the start of field sampling to develop the method and to obtain a sense for how the YSI probe performs under a variety of conditions likely to be encountered at the Moab site (e.g., high salinity, effect of pH, calibration drift). ESL results using the YSI probe were also compared to the Hach spectrophotometric method. The preliminary report, *Preliminary Comparison of Ammonia Determinations with Ion Selective Electrode and Spectrophotometric Methods*, is presented in Appendix C.

4.10.2.2 Ammonia Sampling Results

Field measurements were performed approximately every 100 to 300 ft along the river and sandbar area beginning at about 1,000 ft downgradient from the Area C habitat (Figure 4–23) and progressing upstream to Moab Wash. The sample locations and results of the shallow ground water and river water sampling are summarized in Table 4–17. All the field measurement results were obtained from unfiltered samples. Samples returned to the ESL for ammonia analyses were performed on both unfiltered and filtered samples for comparison. ESL analytical results are also summarized in Table 4–17.

Results of the filtered ammonia laboratory measurements from the August 2002 ground water sampling are posted in Figure 4–24. Ammonia values at all the test pit locations are elevated (nominally 300 to 1,000 mg/L total as N) and consistent with previous sampling conducted in December 2001 (*Chemistry of Ground Water in the Colorado River Sandbar Area*, DOE 2002b). Also posted in Figure 4–24 are the results of filtered laboratory analyses for the surface water samples. Ammonia concentrations in all the river water measurements are consistent with background values except those from a small backwater area identified approximately 1,000 ft down river from Area C (location 0370, Figure 4–24).

A comparison between the unfiltered and the filtered laboratory results for the ground water samples is presented in Figure 4–25. Good agreement between the two Hach spectrophotometric measurements is evidenced by the high coefficient of determination (r^2) value (0.98) shown in the linear regression equation. A similar agreement is obtained between the unfiltered field result (YSI ion selective electrode probe) and the filtered laboratory result (Hach spectrophotometric) as evidenced by the 0.93 coefficient of determination value (Figure 4–26). The linear regression equation indicates the field result, on average, overpredicts the laboratory results by less than 15 percent.

A comparison of ammonia results obtained on filtered samples analyzed in the ESL by the Hach spectrophotometric method for the nine locations where shallow ground water was collected at the same locations in December 2001 and August 2002 is presented in Table 4–18. The daily average river flow during the December 3, 2001, sampling event was approximately 2,960 cfs (USGS station 09180500 near Cisco, Utah), which is 730 cfs more than the 2,230 cfs flow for the August 7, 2002, sampling event. The lower river stage is apparent in the greater sample depths required for the August 2002 event. Even so, the ammonia concentrations are consistent between the two sampling events, as shown in the average coefficient of variation (CV) of 16 percent.

Table 4–17. Field and Laboratory Results From the August 2002 Sampling in the Sandbar Area

Sample	Date Obtained	Field (unfiltered)							Laboratory		Description
		Temp (°C)	pH	ORP (mV)	COND (µS/cm)	NH4 as N (mg/L) probe	NH3 as N (mg/L) probe	Total Ammonia as N (mg/L) probe	Total Ammonia as N (mg/L) unfiltered Hach	Total Ammonia as N (mg/L) filtered Hach	
River Samples											
MOA 370	8/7/2002	32.4	7.5	216	93,560	250	5.98	256.0	27.5	28.5	still backwater, 10 inches deep
MOA 371	8/7/2002	27.3	8.32	109	1,495	0.169	0.027	0.2	<1	<1	flowing river, 4 inches deep
MOA 372	8/7/2002	33.4	8.91	141	1,594	0.384	0.32	0.7	1.5	<1	12 ft from shore, 3 inches deep
MOA 373	8/7/2002	28.3	8.42	152	1,608	0.62	0.115	0.7	<1	<1	15 ft from shore, 6 inches deep
MOA 374	8/7/2002	26.7	8.42	157	1,505	0.231	0.039	0.3	<1	<1	8 ft from shore, 4 inches deep, mod flow
MOA 375	8/7/2002	26	8.38	162	1,500	0.176	0.026	0.2	<1	<1	20 ft from shore, still water, mod flow, 4 inches deep, cobble with algae, sandy bottom
MOA 376	8/7/2002	26.7	8.47	163	1,501	0.195	0.037	0.2	<1	<1	6 ft from shore, 4 inches deep, quiet, sandy to clayey with algae
MOA 377	8/7/2002	26.1	8.43	124	1,513	0.222	0.036	0.2	<1	1.5	3 ft from shore, 4 inches deep, quiet, silty with algae
MOA 378	8/7/2002	25.7	8.38	142	1,495	0.15	0.022	0.2	<1	<1	3 ft from shore, 4 inches deep, moving water, algae covered cobble
MOA 379	8/7/2002	25.8	8.39	150	1,493	0.139	0.02	0.2	<1	<1	3 ft from shore, 4 inches deep, quiet water, cobble with algae, some silt
MOA 380	8/7/2002	26.3	8.43	158	1,512	0.143	0.024	0.2	<1	<1	3 ft from shore, 6 inches deep, good flow, silty bottom, algae with pebbles
Pit samples											
MOA 303	8/8/2002	not enough water for probe reading							390	430	100 ft west of shore, 3.5 ft deep, 2 ft west of cutbank in red sand, minimal H_2O
MOA 306	8/8/2002	22.7	6.77	189	17,230	541	1.481	542.5	450	460	60 ft west of shore, 2.5 ft deep, 6 ft east of cutbank
MOA 308	8/8/2002								490	490	100 ft west of shore, 3.5 ft deep, 6 ft east of cutbank, min water, no readings
MOA 312	8/8/2002	23.6	6.71	201	11,000	448	1.154	449.2	340	360	120 ft west of shore, 2 ft deep, in cutbank
MOA 319	8/8/2002	22.8	6.77	156	18,720	455	1.208	456.2	400	410	100 ft west of shore, 18 inches deep, in cutbank
MOA 328	8/8/2002	22.1	6.92	195	13,790	490	1.77	491.8	560	560	90 ft west of shore, plus 75 ft farther to running water, 2 ft deep, in cutbank
MOA 333	8/8/2002	16.7	6.79	227	28,410	1,073	1.905	1,074.9	960	920	12 ft west of shore, 3.5 ft deep, against cutbank
MOA 334	8/8/2002	16.6	6.78	271	24,750	970	1.658	971.7	760	800	6 ft west of shore, 2 ft deep, against cutbank
MOA 347	8/8/2002	28.4	6.91	178	13,500	523	2.97	526.0	450	450	30 ft west of shore, plus 70 ft to flowing water, 2 ft deep on flat
MOA 376A	8/8/2002	24	6.81	196	17,210	682	2.23	684.2	560	560	8 ft west of 376, 2 ft back from bank, 1.5 ft deep; pit sample
MOA 376B	8/8/2002	26.4	8.2	131	1,635	5	0.445	5.4	8	8	in water, 1 ft east of 376; beneath river bed

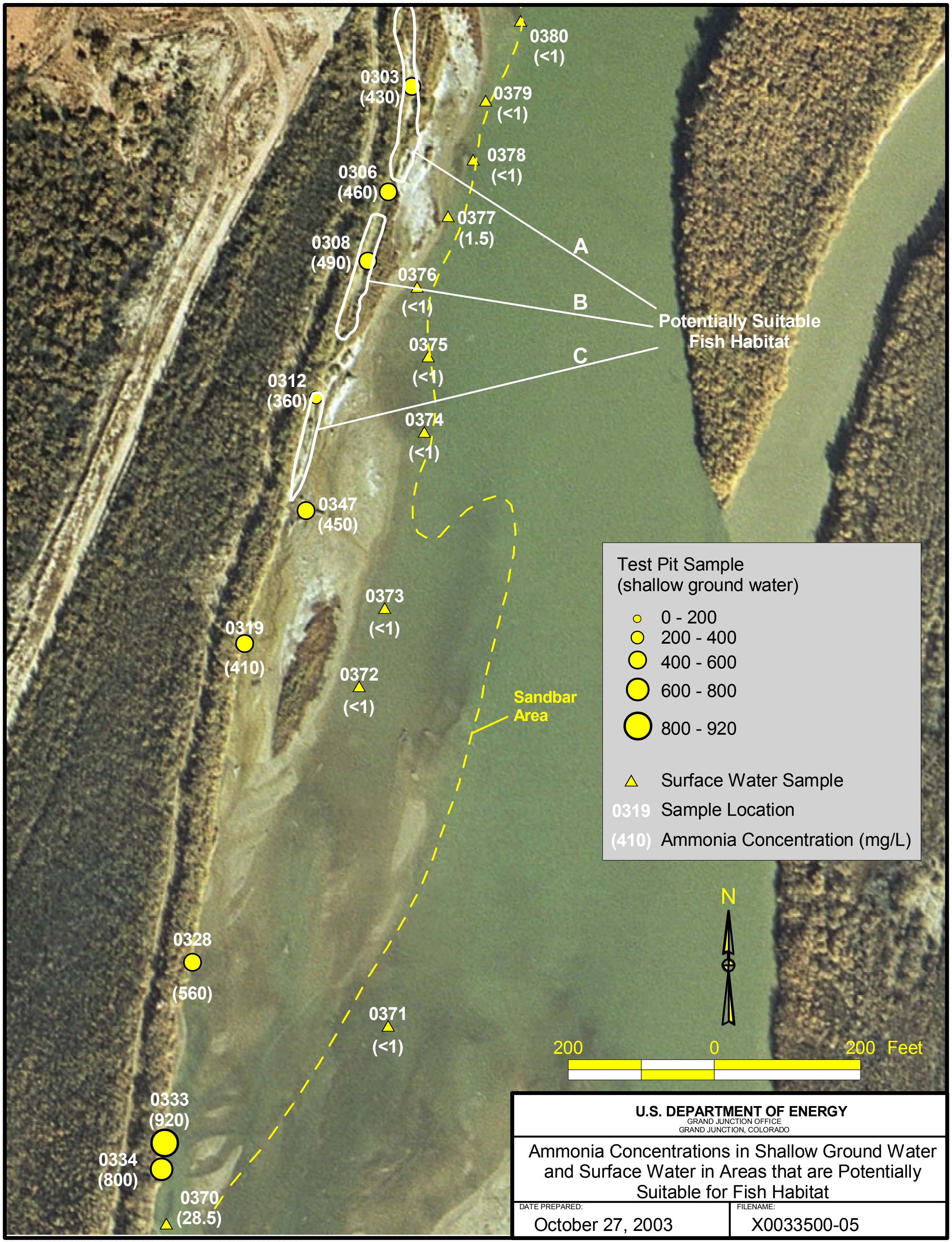

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Figure 4–24. Ammonia (total as N) Concentrations in the Shallow Ground Water and River Water in the Sandbar Area—August 2002

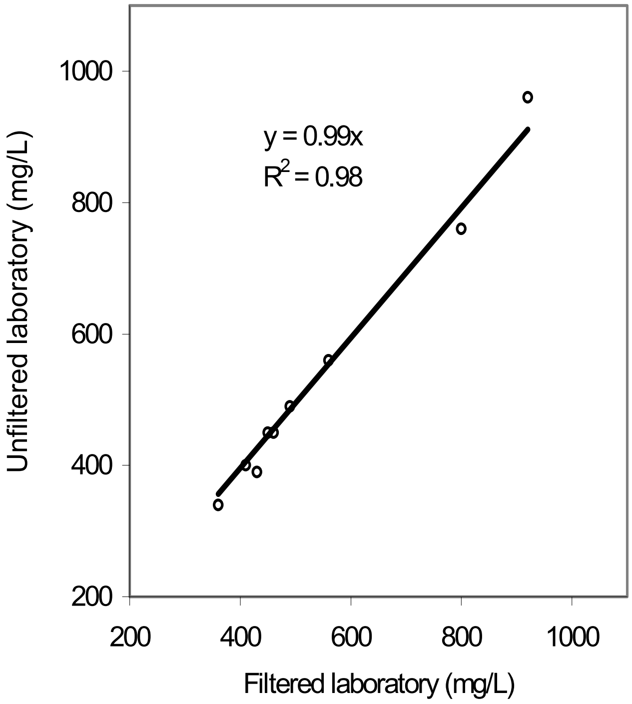

Figure 4–25. Comparison Between Unfiltered and Filtered Laboratory Analyses—Total Ammonia (as N) in Shallow Ground Water

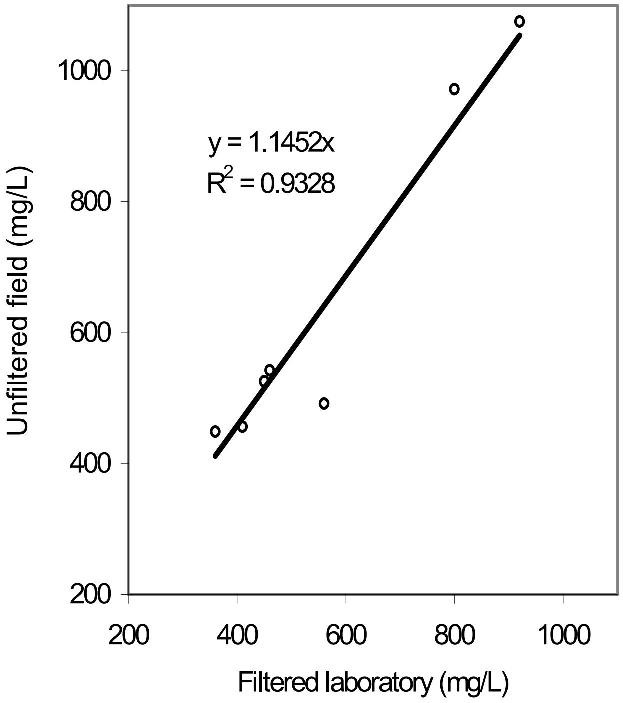

Figure 4–26. Comparison Between Unfiltered Field and Filtered Laboratory Analyses—Total Ammonia (as N) in Shallow Ground Water

Table 4–18. Comparison Between Ammonia Concentrations Measured in December 2001 and August 2002 at the Same Locations from Shallow Test Pits in the Sandbar Area

Sample ID	December 3, 2001		August 7, 2002		CV (%)[a]
	NH_3-N (mg/L)	Sample depth (ft)	NH_3-N (mg/L)	Sample depth (ft)	
MOA 303	505	1.5	430	3.5	11.3
MOA 306	510	1.5	460	2.5	7.3
MOA 308	370	1.5	490	3.5	19.7
MOA 312	320	1.5	360	2	8.3
MOA 319	430	1.5	410	1.5	3.4
MOA 328	290	1.5	560	2	44.9
MOA 333	920	1.5	920	2.5	0.00
MOA 334	500	1.5	800	2	32.6
MOA 347	580	1.5	450	2	17.9
				mean	**16.2**

[a] CV = standard deviation/mean

4.10.3 Floodplain Plant Ecology

Saltcedar, or tamarisk (*Tamarix ramosissima*), is the dominant plant growing along the Colorado River floodplain. Tamarisk has become dominant in southwestern riparian systems for several reasons, including the regulation of streams by dams and the use of river water for agriculture. These factors have caused major changes in alluvial systems, such as a decrease in and a loss of fluctuation in the water table and an increase in salinity in both soils and water. Tamarisk, classified as a phreatophyte (a plant that uses ground water), undoubtedly influences the hydrology and toxicology of the ground water plume between the tailings pile and the Colorado River. Tamarisk and other phreatophytes growing between the tailings pile and the river are currently removing water, ammonia, and possibly other contaminants from the aquifer. However, tamarisk stands in areas with the highest ammonia levels appear to be unhealthy and dying.

For purposes of ground water remediation at the Moab site, the existing role of tamarisk can be described as passive phytoremediation. Phytoremediation is the name given to technologies that use plants to clean contaminated sites (EPA 2000). The rate of phytoremediation is likely greatly diminished in the unhealthy tamarisk stands. A better understanding of the factors causing poor health in these stands and the physiological tolerances of tamarisk to ammonia, salinity, and other potential stressors may lead to options for enhancing phytoremediation as a potential remediation alternative.

Plant ecology and ecophysiology of floodplain vegetation at the Moab site were characterized to begin to address (1) influences of floodplain plants on the site water balance and plume dispersion, (2) current and potential role of plants in remediation alternatives for ground water contaminants, and (3) ecological risks associated with plant uptake of plume contaminants. DOE (2003b) presents details regarding the floodplain plant ecology study. Section 4.10.3.1 summarizes a first order approximation of tamarisk transpiration rates at the Moab site to support development of a conceptual water balance for the site.

4.10.3.1 Tamarisk Transpiration

Some of the factors that affect transpiration rates in tamarisk include humidity, temperature, stand density, leaf area index (LAI), soil salinity, and depth to ground water. In one study of tamarisk transpiration rates, the humidity-transpiration relationship for tamarisk was found to be almost perfectly linear; as the humidity rises, transpiration decreases (Anderson 1982). In a greenhouse study of wind and solar radiation effects, temperature was identified as the only significant factor in predicting transpiration rates (Nagler et al. in press). Several studies (e.g., Horton 1976, 1977; Culler et al. 1976) have shown that the relationship between tamarisk stand density and transpiration is linear at lower densities but levels off as stand density increases. There are indications that dense tamarisk stands with a high LAI and strong advective conditions represent the greatest potential for landscape-level water losses. However, highly dense stands may have a surface boundary layer with a vapor pressure deficit that decreases transpiration rates if advective conditions are not present. This finding is not conclusive, as there have been varying results on the correlation between vapor pressure deficit, stomatal response, and transpiration rates (Vandersande et al. 2001; Horton and Clark 2001; Anderson 1982; Busch and Smith 1995). Several studies have shown that tamarisk transpiration rates decrease with increasing soil salinity (Van Hylckama 1970; Hagemeyer and Waisel 1989, Glenn et al. 1998, Vandersande et al. 2001). Conflicting evidence is reported on the effect of water table depth on water use by tamarisk. Busch et al. (1992) showed decreasing transpiration rates with increasing water table depth, and Weeks et al. (1987) showed a constant rate. Competition with native riparian species also plays a role in water use (Cleverly et al. 1997; Busch et al. 1992, Vandersande et al. 2001, Horton and Clark 2001).

Variations in climate and environment as well as differences in measurement methods contribute to the variability in the literature estimates of water use by tamarisk presented in Table 4–19. Most of these studies have been conducted in locations south of Moab, Utah, including the lower Colorado Basin in California, New Mexico, and Arizona. Most studies have shown that tamarisk water use rates are comparable to those of other riparian vegetation when expressed on a leaf area basis.

Table 4–19. Comparison of Tamarisk Water Use Reported in the Literature

Water Use	Location	Method Used	Reference
1.5−2.1 m/yr	New Mexico and Arizona	Lysimeter	Horton 1976
0.7−0.8 m/yr	Blythe, California	Bowen ratio energy budget	Ball et al. 1994
0.8−1.0 m/yr	NA	NA	Culler et al. 1982
2−11 mm/day, 1.6−1.7m/yr	Blythe, California	Bowen ratio model	Gay and Hartman 1982
6−16 mm/day	Nevada	Stem heat dissipation	Sala et al. 1996
25 mm/day	Tucson, Arizona	Stem heat dissipation	Nagler et al. in press
1.2 g water/g leaf/hr	Rio Grande, New Mexico	Gas exchange cuvette	Anderson 1982
2.3 g water/g leaf/hr	Tucson, Arizona	Water balance	Glenn et al. 1998

NA = information not available
m/yr = meters per year

Range of water use and area for three transpiration zones mapped in July 2001, within the tamarisk-dominated Moab site floodplain, are presented in Table 4–20 and Figure 4–27. The map (Figure 4–27) was generated from a combination of sources: literature on tamarisk transpiration rates (Table 4–20), results of the demonstration of methods for measuring tamarisk growth and stress (DOE 2003d), and a false-color infrared (IR) image of the site produced by the Savannah River Technology Center using data from DOE's MTI satellite-based system developed by Sandia National Laboratories (Figure 4–28).

Table 4–20. Ranges of Water Use and Area of Three Transpiration Zones Mapped in the Tamarisk-Dominated Moab Site Floodplain

Zone	Water Use Range (meters/year)	Zone Area (acres)
1	2.5–5.0	32.6
2	1.0–2.5	20.5
3	0.0–1.0	33.4

False-color IR images provide a valuable means for differentiating healthy growing vegetation from unhealthy or dormant vegetation. False-color images block out light reflectance in the green wave band region of the visible spectrum and intensify the near-infrared bands. Healthy, growing vegetation appears bright red. Unhealthy vegetation appears as shades of gray and blue. In the September 2001 image of the Moab site (Figure 4–28), the degree or intensity of reddish tones is an indication of the vigor of tamarisk growth. Infrared is a measure of temperature, not visible light. The spatial heterogeneity of red patterns in the image corresponds to areas with high and low latent heat flux resulting from varying rates of evaporation and transpiration. The bright red color of the island in the Colorado River and along the shoreline indicate healthy tamarisk stands with relatively high transpiration rates. Dark gray areas along the Moab site floodplain and to the east in the Matheson Wetlands Preserve indicate unhealthy or dormant tamarisk stands with relatively low transpiration rates.

The three transpiration zones were delineated by hand and are based on the degree of red color in the September 2001 false-color IR image. Zone 1 corresponds to the most intense red along the Colorado River, Zone 2 corresponds to the less intense or faded red areas, and Zone 3 corresponds to gray areas with only a hint of red. Ranges of annual transpiration rates, obtained from the literature on tamarisk water use (Table 4–19), were assigned to each of the three zones (Table 4–20). The highest water-use value from the literature search, up to 25 mm/day, is from the Tucson, Arizona, area that has a 365-day growing season and was assumed to be twice the potential water use by the tamarisk stand growing in the Moab area.

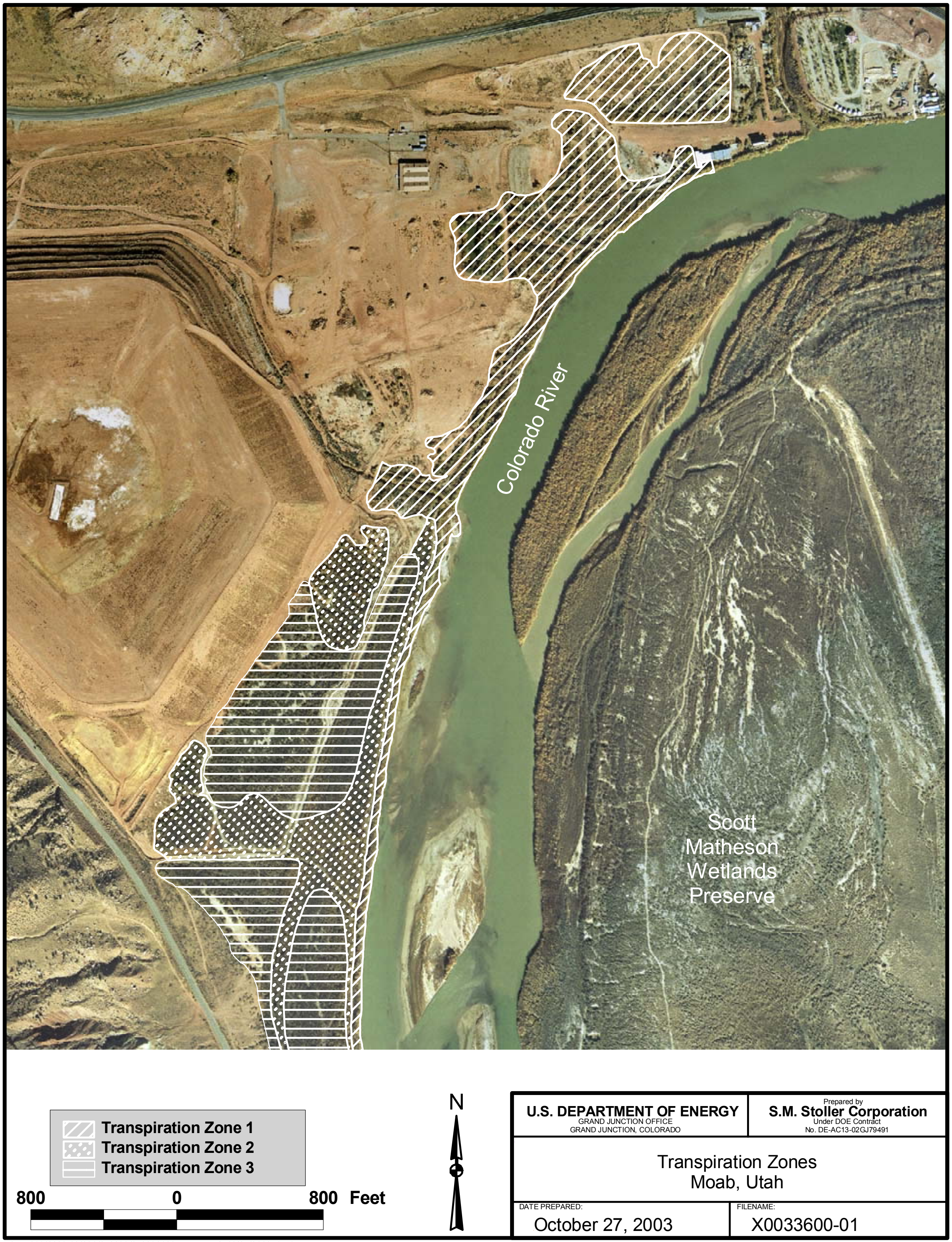

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Figure 4–27. Preliminary Map of Transpiration Zones

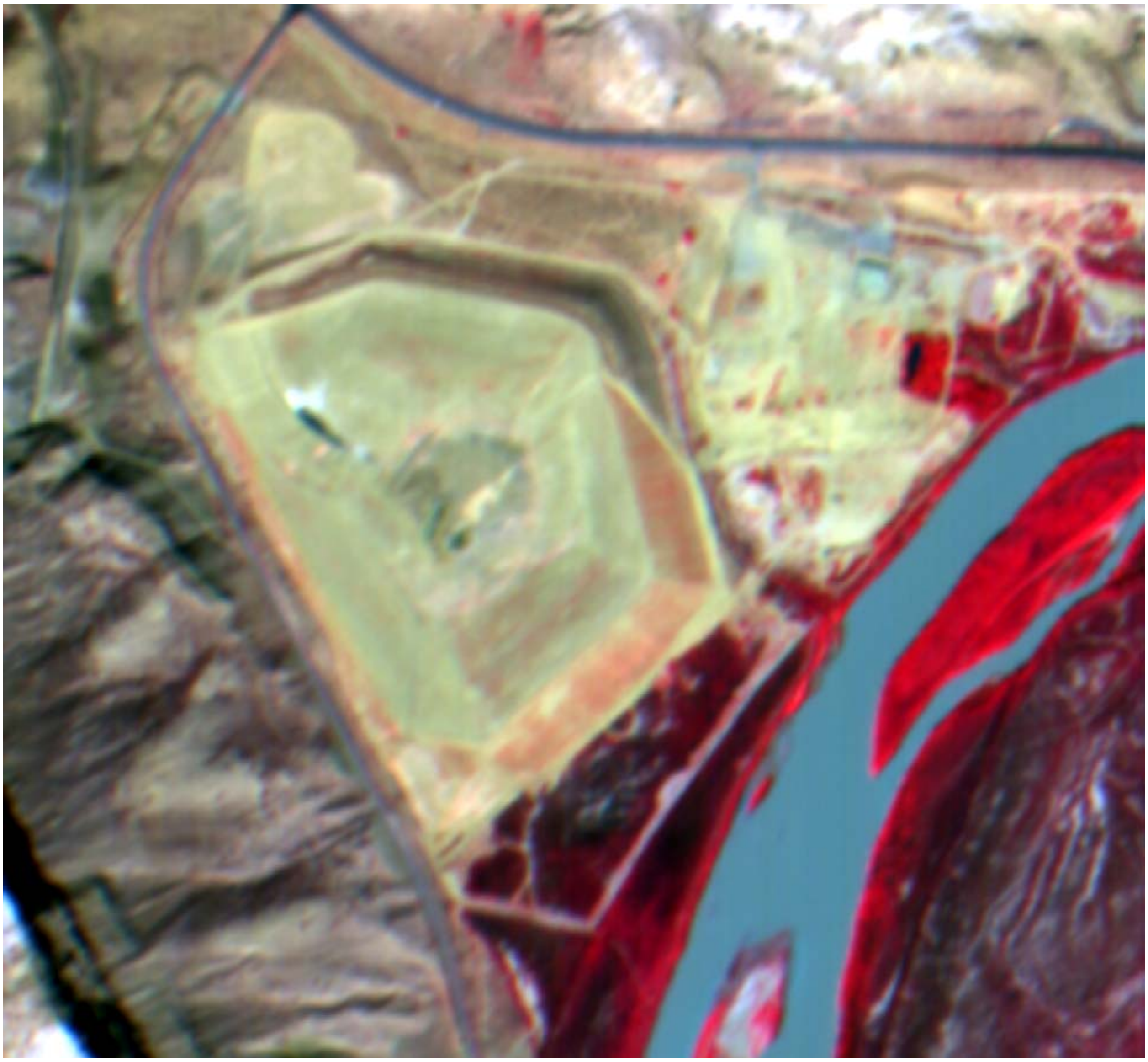

Figure 4–28. False-Color Infrared Image of the Moab Site and Floodplain Derived from DOE MTI Satellite Data Obtained in September 2001

4.11 Summary of DOE Investigations and Reports

DOE reports prepared from data collected in 2001 and 2002 that are most pertinent to support the site conceptual model and for the development of the ground water compliance strategy for the Moab site are listed in Table 4–21. The reports are listed in groups according to the primary focus of the investigation. Reports that are calculation sets are also included in Appendix D.

Table 4–21. Summary of Pertinent DOE Reports for Data Collected in 2001 and 2002

Reference	Description
Tailings	
Determination of Distribution Ratios, December 2002.	Laboratory batch tests on background alluvial samples to determine amount of uranium and ammonia that sorbs to matrix. Evaluate tendency for contaminants to be retarded by the aquifer.
Determination of Subpile Soil Concentrations, January 2003	Laboratory 5% acid leach test of soil samples collected beneath tailings pile. Analyzed for ammonia, arsenic, iron, manganese, molybdenum, selenium, sulfate, uranium, and vanadium to evaluate the potential for these soils to act as a long-term source of ground water contamination.
Tailings Seepage, January 2003.	Estimate of the volume and rate of seepage of pore water from the tailings pile. Long-term, steady-state drainage (gravity), and transient flux from consolidation using HYDRUS 1D model.
Ground Water	
Characterization of Groundwater Brine Zones at the Moab Site (Phase I), June 2002.	Evaluate potential for intrusion of the lower brine zone into the shallow freshwater system from pumping a ground water remediation extraction well. Results of 2 short-term aquifer tests to determine sustainable pumping rate for an extraction well and to determine aquifer parameters.
Lithologic, Well Construction, and Field Sampling Results from the 2002 Field Investigation, October 2002.	Boring logs and construction information for wells installed using the sonic drilling method during the 2002 field campaign. Includes radiocarbon age date analysis on wood fragment, and field conductance measurements on ground water grab samples.
Soil Conductivity Investigation Results, November 2002.	Evaluates nature and extent of the interface between the overlying freshwater zone and the deeper brine system.
Alluvial Aquifer Vibrating Wire Piezometers, December 2002.	Pressure profiles determined at three nested locations in the alluvial aquifer adjacent to the site. Presents converted pressure readings to hydraulic head expressed in feet of freshwater.
Aquifer Test Data Analyses (Phase II, Part 2), January 2003.	Results of 2 long-term and 3 short-term aquifer tests. Determine effect of the brine upconing during long-term pumping test and aquifer parameters from short-term test.
Surface Water	
Chemistry of Ground Water in the Colorado River Sandbar Area, January 2002.	Investigated interactions between the alluvial aquifer and the surface water in the potentially critical fish habitat area adjacent to the site. Ground water samples collected from 57 shallow test pits analyzed for uranium, ammonia, sulfate, chloride and field parameters (ORP, dissolved oxygen, and pH).
Data Validation Reports	
December 2001 Water Sampling.	First DOE sampling event. Monitor well and surface water sample locations similar to previous trustee contractor (SMI Inc.).
March 2002 Water Sampling.	Similar to first round of sampling. Added volatile organic compounds screening analysis on water sample collected from well screened in shallow aquifer downgradient near toe of pile. Added new surface water sample location (201) approximately 6,000 ft farther downgradient from site.
May 2002 Water Sampling.	Added organic analyses (semivolatiles, volatiles, and PCBs) for tailings pore fluid samples and for ground water and soil samples collected at five new monitor wells installed in millsite area. Isotopic uranium analyzed for all surface and ground water samples. Additional surface water samples collected from bottom of main channel. Remaining sampling similar to previous rounds.
June through September 2002 Water Sampling.	Includes grab samples of tailings pore fluids and ground water collected in June, July, and August 2002 during the field campaign and analyzed by the ESL. Expanded quarterly sampling in August and September 2002 to include new wells installed during the field campaign and additional historical wells not previously sampled by DOE. Expanded analyte list to include radon-222, aluminum, boron, beryllium, fluoride, lithium, phosphate, silica, and total organic carbon. Most comprehensive analyte list to date.
December 2002 Water Sampling.	Decreased analyte list and increased number of monitor wells sampled based on evaluation of potential contaminants of concern and results of previous analyses. Added new surface water sample location upgradient of site (205) and adjacent to site (204). Most comprehensive sampling to date.

End of current text

5.0 Site Characteristics

5.1 Geology

5.1.1 Regional Geology and Structural Setting

The Moab site is in the fold and fault belt in the northern part of the ancestral Paradox Basin (Figure 5–1). The fold and fault belt is characterized by northwest-striking salt-cored anticlines and synclines that are cut in places by normal faults and joints that also mainly strike northwest. The broad northwest-striking Colorado lineament that contains basement wrench faults of left-lateral displacement transects the Paradox Basin. The site area lies along the most significant segment of the Colorado lineament, a narrow belt of basement faults called the Cataract lineament (Figure 5–1) that mainly follows the course of the Colorado River (Stevenson and Baars 1987). The following geologic history of the Paradox Basin and fold and fault belt development is summarized from Doelling et al. (2002).

The Paradox Basin formed adjacent to the ancestral Uncompahgre Uplift during Pennsylvanian to Late Triassic time. The basin was asymmetrical and its deepest (northern) part was just southwest of the Uncompahgre Uplift. Clastic, carbonate, and evaporite sediments were deposited in the basin. Most evaporite deposits, which contained large amounts of halite, were deposited in the Paradox Formation during Middle Pennsylvanian time. Discontinuities existed in the floor of the basin in the form of northwest-striking faults of pre-Pennsylvanian age. The low-density evaporite deposits flowed toward these discontinuities where they thickened to form northwest-striking elongate salt diapirs. The thickness of evaporites in these diapirs is known to reach as much as 15,000 ft (Baars and Doelling 1987). During salt movement, sediments deposited in the basin from Late Pennsylvanian through Triassic time were generally thinned or were not deposited across the crests of the salt diapirs. Basins called rim synclines formed between the salt diapirs as the salt flowed toward the diapirs. Thicker deposits of Late Pennsylvanian to Late Triassic sediments filled these basins.

After the end of Paradox Basin sedimentation, salt continued to move during Jurassic and Cretaceous time, affecting the thickness of strata deposited over and near the salt diapirs. Regional compression during the Laramide orogeny of Late Cretaceous to early Tertiary time formed broad northwest-striking anticlines and synclines. Anticlines were superimposed over the salt diapirs, and synclines formed between the anticlines. This compression formed the Moab Valley salt-cored anticline, where the Moab site is located, and the adjacent Courthouse syncline and Kings Bottom syncline structures to the northeast and southwest, respectively (Figure 5–1).

After the Laramide folding, northwest-striking faults, such as the Moab Fault (Figure 5–2), cut the folds. This Tertiary-period extensional faulting may be related to one or more of the following causes: possible reactivation of subsurface faults of pre-Pennsylvanian age, regional extension after the Laramide compressional folding, and epeirogenic uplift of the Colorado Plateau during the late Tertiary. During the epeirogenic uplift of the Colorado Plateau for approximately the past 10 million years, subsequent erosion has cut deeply into the Cretaceous and older sedimentary rocks and has carved the extensive canyons of the present regional physiographic setting. Physiographically, the Moab site is in the northern part of the Colorado Plateau province, in the north part of the Canyonlands section.

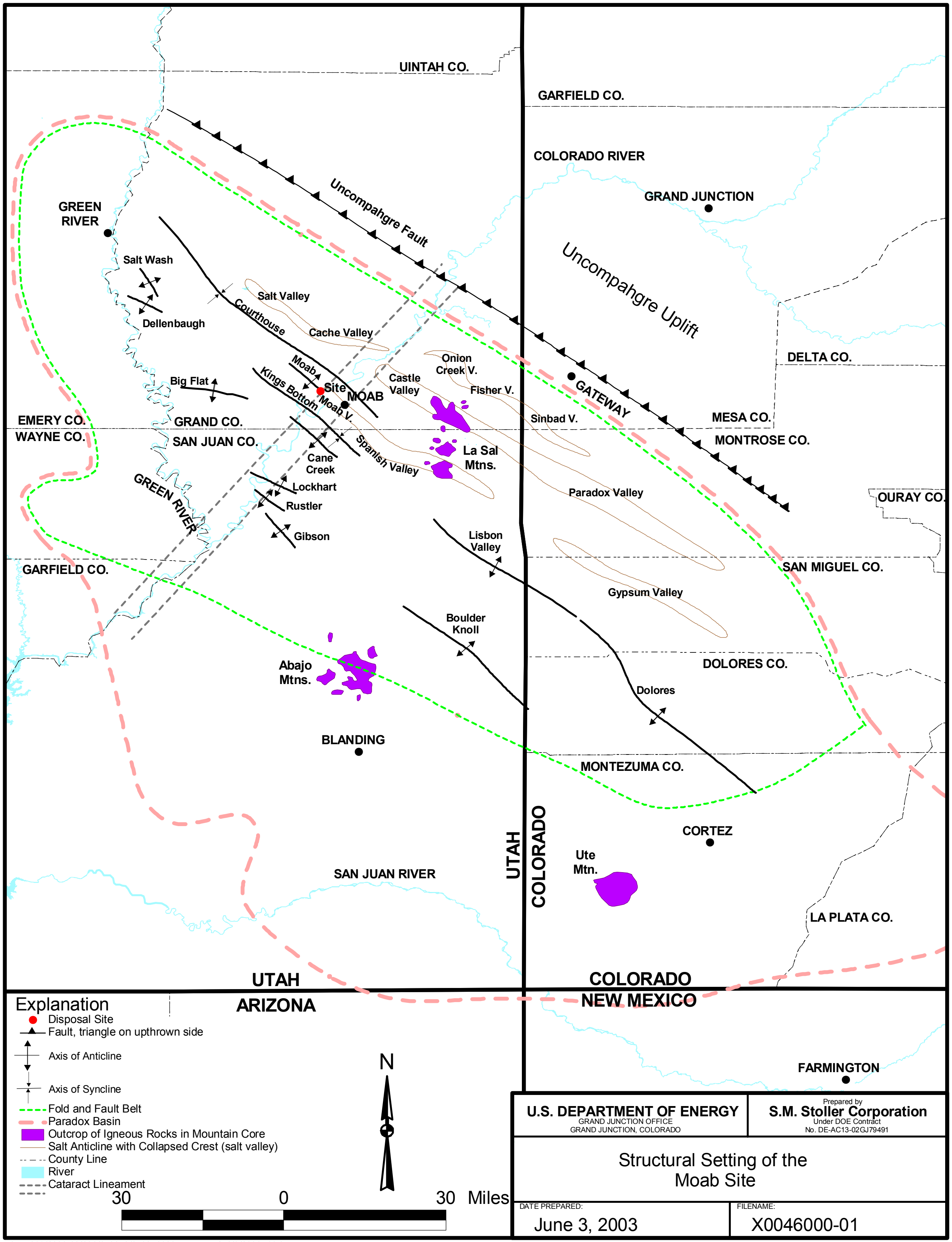

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Figure 5–1. Geologic and Structural Setting

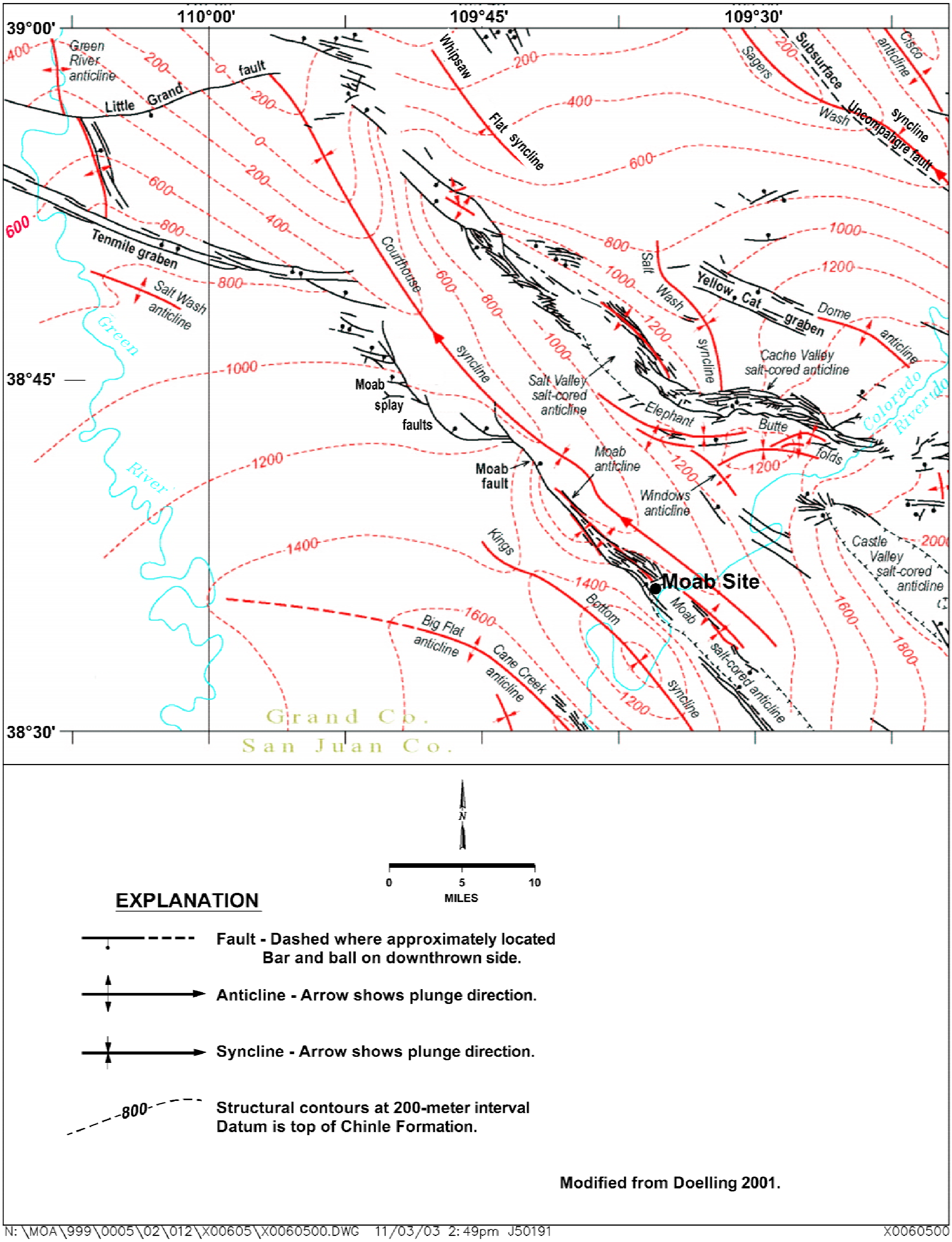


Figure 5–2. Faults and Folds in the Moab Area

The late Tertiary erosion allowed ground water to locally reach the upper parts of the salt diapirs through fractures and joints in the anticlinal folds. The resulting dissolution during late Tertiary and Quaternary time (and to the present) caused local areas of collapse, tilting, faulting, and subsidence of the overlying strata along the salt-cored anticlines. The degree of breaching (or unzipping) of the salt-cored anticlines in this part of the Colorado Plateau largely reflects the amount of ground water that has been available for dissolution. Ground water dissolves the salt and carries it away, leaving the insoluble part of the Paradox Formation as residue, called cap rock, on top of the leached salt diapirs.

5.1.2 Site Geology

The Moab site is located at the northwest end of Moab Valley, which formed during late Tertiary and Quaternary time by salt-dissolution-induced subsidence along the axis of the Moab-Spanish Valley salt-cored anticline. The site is situated at the mouth of Moab Canyon where Moab Wash, an ephemeral drainage passing through the site, follows Moab Canyon northwestward and also is approximately along the trace of the Moab fault. Cutting across the structural grain of this area, the Colorado River flows southward along the east edge of the site. At the northeast and southwest edges of Moab Valley, the Colorado River flows in deeply incised bedrock canyons cut by the superimposed river during the past several million years. The Colorado River flows southward out of Moab Valley through the Portal, the 1,000-ft sandstone cliffs flanking the river canyon mouth. The steep slope southwest of the site flanking Moab Valley rises 1,200 to 1,400 ft to the top of Poison Spider Mesa, capped by sandstones of the Wingate and Kayenta Formations. Just north of the site, north of US-191 and at the north end of Moab Valley, is a steep slope that rises approximately 600 ft and consists of highly fractured and faulted sandstones of the Wingate, Kayenta, and Navajo Formations (composing the Glen Canyon Group of Jurassic age). Dips of bedrock on this slope express the form of the Moab anticline, which is the northwest extension of the Moab Valley salt-cored anticline.

Plate 2 is a detailed geologic map of the Moab site and nearby area. This map is part of, and only slightly modified from, the excellent geologic mapping of the Moab 7.5-minute quadrangle conducted recently by Doelling et al. (2002) for the Utah Geological Survey. Seven cross-sections were constructed through the site to include the lithologic information from the 14 boreholes drilled during the 2002 field investigation (Section 4.0) and from selected boreholes from earlier drilling (Section 3.0). The lines of these cross sections (A-A’, B-B’, C-C’, D-D’, E-E’, F-F’, and G-G’) are shown in Plate 2, and the individual sections are shown in Plates 3 through 9, respectively. Three of the cross sections (B-B’, C-C’, and E-E’) were used in the presentation of the fence diagram through the site shown in Plate 10. A map of the bedrock geology of the site as it would appear if the Quaternary material were removed is shown as the subcrop map in Plate 11.

5.1.2.1 Bedrock Formations

Rocks exposed and in subcrop in the site area, as shown in Plates 2 and 11, respectively, range in age from Middle Pennsylvanian to Middle Jurassic. These rock formations, from oldest to youngest, and their occurrence around the site are described in the following section. The geologic structures that affect these formations in the site area were described previously in Section 5.1.1.

Paradox and Honaker Trail Formations

The oldest rocks deposited in Middle and Late Pennsylvanian time in the site area include the Paradox and Honaker Trail Formations, respectively, which comprise the upper two (of three) units in the Hermosa Group that were deposited in the Paradox Basin. The Paradox Formation was deposited in a periodically restricted part of the basin and consists of cyclically bedded evaporites, dolomite, organic-rich carbonaceous shale, and siliciclastic shale and siltstone. Because of lateral and vertical flowage of the low-density evaporite deposits as their thickness built up (described in Section 5.1.1), the thickness of the Paradox Formation is highly variable. Along Moab Valley, the Paradox Formation thickness is estimated to be at least 9,000 ft (Doelling et al. 2002). Before movement of the evaporites, the original depositional thickness of the Paradox Formation in this part of the Paradox Basin is estimated at about 5,000 ft (Hite and Lohman 1973).

No outcrops of Paradox Formation are present in the immediate site area; however, the formation crops out about 1.5 miles southeast of the site across the Colorado River along the southwest and northeast margins of Moab Valley. Those grayish white outcrops consist of cap rock, which is the insoluble residue of the formation consisting of contorted masses of gypsiferous mudstone, black shale, sucrosic gypsum, and fragments of limestone, sandstone, and dolomite. Although the site contains no outcrops of Paradox Formation, cap rock covering the undissolved evaporites of the formation is believed to be present immediately below the Quaternary material in much of the site, as shown in the subcrop map in Plate 11.

An area of upward flow of the Paradox Formation is inferred (not confirmed by deep boreholes) along the southwest flank of Moab Valley. In this area, shown in cross sections A-A’, D-D’, and E-E’, the Paradox Formation is inferred to have bulged upward, nearly to the surface in E-E’, similar to the Paradox cap rock southeast of the site. Highly saline ground water at a shallow depth from monitor wells in the area of the south end of cross section E-E’ provides evidence that the saline Paradox Formation evaporites are at a shallow depth, and active dissolution is taking place. Northwest of the south end of cross-section E-E’, the depth to the Paradox Formation cap rock becomes progressively greater, as inferred in cross-sections D-D’ and A-A’.

None of the 14 boreholes drilled during the 2002 field investigation and none of the earlier boreholes are known to have penetrated cap rock of the Paradox Formation beneath the site. The thickness of the cap rock is at least 400 ft, as found in the Great Lakes Carbon Corporation No. 1 well drilled in 1943 in Moab Valley about 1.5 miles southeast of the site. Doelling et al. (2002) estimated that the cap rock may be as much as 700 ft thick in Moab Valley.

The Honaker Trail Formation consists of interbedded sandstone, limestone, and siltstone that crop out as ledges on the steep slope leading up to Poison Spider Mesa west and south of the tailings pile. Along this slope, beds of the Honaker Trail dip toward the southwest at 15 to 25 degrees toward the Kings Bottom syncline (Plate 2). Doelling et al. (2002) estimated that only as much as the upper 25 percent, or 600 to 700 ft, of the Honaker Trail are exposed on the Poison Spider Mesa slope; salt movement has greatly reduced the exposed thickness. The Honaker Trail Formation was deposited in shallow marine shelf and nearshore environments. None of the 14 boreholes drilled in 2002 and none of the earlier boreholes are known to have penetrated the Honaker Trail Formation beneath the site. The subsurface presence of the Honaker Trail at the site is inferred in the subcrop map in Plate 11.

Cutler, Moenkopi, and Chinle Formations

The Cutler Formation consists of interbedded red to red-orange, mostly fine-grained, subarkosic to quartzose, eolian sandstone; red to purple arkosic sandstone; and conglomeratic fluvial sandstone. Up to 600 ft of Cutler is exposed south and west of the site on the slope of Poison Spider Mesa. Cutler thickness is much greater to the southwest in the Kings Bottom syncline. Thickness variations are due to salt tectonic activity during the Permian; Cutler is commonly missing over the tops of salt diapirs. The formation was deposited in a transitional zone between an alluvial fan environment along the southwest flank of the ancestral Uncompahgre highland, an eolian environment to the south and southwest, and a shallow marine environment to the west-northwest (Doelling et al. 2002).

The Moenkopi Formation consists of interbedded siltstone, fine-grained sandstone, and mudstone. The overall color is light to dark brown, or “chocolate brown.” Ripple marks are characteristic of the formation in thin-bedded, fine-grained sandstones. Outcrops are visible south and west of the tailings pile on the upper and lower slopes of Poison Spider Mesa. Outcrops on the lower slope have been displaced downward by the west branch of the Moab Fault. Moenkopi thickness variations are due to paleo-erosion at the angular unconformity below the Chinle Formation, and the Moenkopi is missing over the top of the Moab salt-cored anticline (Doelling et al. 2002). Red beds of this formation were deposited in fluvial, mudflat, and shallow water environments.

The Chinle Formation forms gray-red to red-brown, ledgy slopes. It consists of interbedded fluvial sandstone, mudstone, siltstone, and pebble conglomerate. Mottling is common and indicates pedogenic alteration. A middle ledge-forming interval contains conglomeratic sandstone with fossil wood and vertebrate fossil fragments; a distinctive, massive conglomeratic sandstone ledge at the top of this interval is coated with brown desert varnish and is called the Black Ledge. Outcrops of Chinle are on the lower and upper slopes of Poison Spider Mesa south of the tailings pile (the lower slope outcrops have been displaced downward by the west branch of the Moab Fault) and northeast of the site along the lower slope of Moab Valley on both sides of the mouth of Courthouse Wash. Chinle thicknesses vary considerably and thin over the Moab anticline. Deposition of the formation was in alluvial channel and floodplain environments; eolian environments developed near the end (Doelling et al. 2002).

Wingate Sandstone, Kayenta Formation, and Navajo Sandstone

The Wingate Sandstone is composed of gray-orange to pale red-brown, very fine to fine-grained sandstone, which is quartzose subarkose. Sand grains are moderately to well-sorted and frosted, suggesting eolian transportation. The sandstone forms massive ledges and cliffs that are streaked and stained with dark brown desert varnish. In the site area, the Wingate forms a prominent gray-pink to red-brown smooth cliff along the upper slope of Poison Spider Mesa, and forms a wall along the northeast side of Moab Valley and at the mouth of Courthouse Wash where Doelling et al. (2002) determined that the formation is 250 ft thick. The Wingate is faulted and highly fractured near the Moab anticlinal axis as it plunges southeastward into Moab Valley.

The contact of the Wingate and overlying Kayenta Formation is where vertical cliffs end and thick ledges of Kayenta begin. Deposition of the Wingate was in desert environments where eolian dune and interdune sediments accumulated.

The Kayenta Formation consists mainly of very fine grained to medium-grained sandstone that forms dark brownish-red ledgy cliffs. The moderately to well-indurated sandstone is a resistant unit. The character of the formation consists of lenses of fluvial sandstone and siltstone interbedded with lesser fluvial conglomerate and eolian sandstone. In the site area, the Kayenta caps Poison Spider Mesa to the south; north of the site, the formation crops out along the edge of Moab Valley near the mouth of Cottonwood Wash, where Doelling et al. (2002) measured a formation thickness of 293 ft. The formation was deposited mainly in sandy fluvial systems with eolian environments present near the top of the formation.

The Navajo Sandstone is light gray to red-orange and consists of fine-grained quartz sand that is well-sorted, rounded, and frosted. The sandstone is friable to moderately indurated. The lower part of the formation consists of interbedded flat- and cross-bedded sandstone, and the upper part is large-scale cross-bedded sandstone that weathers to rounded cliffs and domes. Exposed at the site north of US-191, the Navajo Sandstone forms the northwest end of Moab Valley and dips moderately (about 50 degrees) southwest along the southwest flank of the Moab anticline. Thickness of the Navajo Sandstone varies and is approximately 400 ft across the crest of the Moab salt-cored anticline (Doelling et al. 2002). The Navajo was deposited mainly in an eolian environment.

Carmel Formation, Entrada Sandstone, and Curtis Formation

One member of each of the Carmel and Entrada Sandstone Formations is present in the northwest end of the site area in the subsurface just north of the Moab Fault in the lower end of Moab Canyon. Both formations are part of the San Rafael Group. The Dewey Bridge Member, earlier classified as part of the Entrada Sandstone, is considered from recent work to be an eastern extension of the Carmel Formation (Doelling et al. 2002). The Dewey Bridge Member unconformably overlies the Navajo and consists of red-brown, silty, mostly fine- to medium-grained sandstone with irregular, contorted bedding. The Slick Rock Member, which is the only member of the Entrada Sandstone in the site area, consists of thick-bedded, red-orange to brown, very fine to fine-grained, eolian sandstone. The Slick Rock sandstones are well-fractured in the subsurface along the Moab Fault zone.

5.1.2.2 Quaternary Deposits

Quaternary deposits in the site area may be classified as alluvial, eolian, or mass-movement in origin. Except for the alluvial deposits, most Quaternary deposits are relatively thin. Because of the subsidence caused by removal of salt from the underlying Moab salt-cored anticline, alluvium deposited mainly from the ancestral Colorado River has accumulated to a thickness of as much as 450 to 500 ft beneath the site in Moab Valley. The subsiding Moab Valley has acted as a sump to catch Colorado River alluvium for much of Pleistocene time since erosion has begun “unzipping” the Moab salt-cored anticline and exposing the salt to dissolution by ground water and surface water.

Alluvium

The thick basin-fill alluvial deposits consist mainly of coarse gravelly sand, with minor silt and clay. Boulders as large as 1 to 2 ft in diameter, composed of resistant igneous and metamorphic rocks representing the upper Colorado River drainage, are common in the alluvium; rare thin interbeds of mainly fine-grained sand in the alluvial sequence represent episodes of eolian deposition.

In the northwest end of Moab Valley at the mouth of Moab Canyon, the Colorado River alluvial deposits are mixed with and interlayered with generally finer-grained alluvium and detritus that has traveled down Moab Wash. This alluvium lacks the rounded, resistant, exotic rock types characteristic of the upper Colorado River drainage and consists mainly of locally derived sandstone, limestone, and chert.

Overlying the coarse alluvial deposits in the immediate site area in Moab Valley adjacent to the Colorado River is finer-grained alluvium of Holocene age composed mainly of sand, silt, clay, and minor lenses of gravel; this modern alluvium of the Colorado River covers much of the site area outside the tailings pile and is approximately 15 to 40 ft thick (Plate 10).

South and west of the tailings pile, eolian and mass-movement deposits (consisting mainly of talus) cover at least half of the upper and lower slopes of Poison Spider Mesa. The eolian sand deposits are Holocene in age, are generally light red-orange to light red-brown, and consist of fine- to medium-grained quartzose sand and silt. These sand deposits have accumulated on the lee slope of Poison Spider Mesa; at the base of the mesa in lower Moab Canyon, the sand is thicker and forms dunes in some places. Talus deposits consist of gravity-induced rockfall blocks mixed with slopewash on steep slopes below cliffs; some talus also is present just north of the site on the steep sandstone slopes north of US-191.

5.1.2.3 Geologic Hazards

Swelling clay (montmorillonite) is present in the site area in the Moenkopi and Chinle Formations. These formations crop out along the west edge of the site in the lower slopes of Poison Spider Mesa. These bentonite-derived clays are capable of absorbing large amounts of water, accounting for the shrinking and swelling characteristics of the soils derived from these formations.

Piping and rapid erosion may occur in fine-grained soils and unconsolidated fine-grained sediments at the site, such as the older alluvium deposited along the ephemeral stream channel of Moab Wash. The piping can occur when ground water flows into permeable, noncohesive layers, removes fine sediments, and exits where the layer reaches the surface (Doelling et al. 2002). The void space created is a “pipe” that accelerates erosion.

Active rockfall areas are along the Wingate Sandstone cliff at the top of the slope of Poison Spider Mesa. Rockfall debris may travel down this slope to the southwest margin of the site. Large boulders in the talus deposits on the slopes below the Wingate cliffs and the Black Ledge layer in the Chinle Formation are evidence of previous events.

Seismic and salt dissolution hazards were evaluated for the site area by Woodward-Clyde Federal Services (1996). These hazards consist of the Moab Fault and its capability to rupture the surface of the site, the potential for salt dissolution and collapse at the site, and seismogenic potential of the microearthquake trend along the Colorado River and other Precambrian faults in the site region.

The Moab Fault is one of the longest of several northwest-striking, northeast-dipping normal faults in the fold and fault belt of the Paradox Basin that are associated with northwest-striking salt-cored anticlines. In the vicinity of the site, the Moab Fault consists of two branches—the main Moab Fault and the west branch of the Moab Fault, which is exposed in places west and southwest of the site on the slopes of Poison Spider Mesa. The displacement of the main Moab Fault is as much as 3,100 ft (Foxford et al. 1996) but is considerably less in the site area. Doelling et al. (2002) estimated the displacement on the main Moab Fault where it is last exposed, about 0.5 mile northwest of the site, at about 1,750 ft, and estimated the displacement on the west branch of the Moab Fault at about 600 ft. In the immediate site area, the main fault is not exposed; it is present far beneath the site in rocks older than the Paradox Formation. Thick salt deposits of the Paradox Formation that have diapirically moved upward in the Moab Valley salt-cored anticline have obscured the main fault. The inferred trace of the main fault before salt dissolution passes through the site approximately across the northeast corner of the tailings pile (Doelling et al. 2002).

Investigations of the Moab Fault system (Woodward-Clyde Federal Services 1996) suggested that primary displacement on the fault system occurred before, and is unrelated to, Quaternary dissolution subsidence in Moab-Spanish Valley. No historical macroseismicity has been noted along the Moab Fault, and microseismicity studies have not revealed any earthquakes associated with the fault. Field and subsurface evidence suggest that the Moab Fault is a shallow structure and would not likely be capable of producing significant earthquakes. For these geologic and geophysical reasons, the Moab Fault system is not a capable fault, as defined in 10 CFR 100, "Reactor Site Criteria," and does not pose a significant earthquake or surface-rupture threat to the present tailings pile.

Seismicity in this part of the northern Paradox Basin has a low rate of occurrence with small- to moderate-magnitude earthquakes (Wong and Humphrey 1989). The site area is in Uniform Building Code 1, indicating lowest potential for earthquake damage (International Conference of Building Officials 2000). A concentration of seismicity along the Colorado River south of the site (coinciding with part of the Cataract lineament) related to multiple faults in the Precambrian basement was evaluated in a probabilistic seismic hazard analysis by Woodward-Clyde Federal Services (1996) along with the Moab Fault. On the basis of that analysis, the recommended design peak horizontal acceleration was 0.18g. For a 10,000-year return period for a strong earthquake, this value provides the level of protection equivalent to the extent practicable as specified in 10 CFR 100. This return interval is credible because of the presence in the site area of precariously balanced rocks, which implies that significant earthquakes are rare.

Vertical subsidence rates in the northwest end of Moab Valley in the site area provide an estimate of the amount of collapse that could be expected from continued salt dissolution beneath the site. Aggradation of Colorado River alluvial material has essentially kept pace with subsidence caused by salt dissolution in Moab Valley, as evidenced by the large thickness (up to 500 ft) of river alluvium that has been deposited (as if in a sump) beneath Moab Valley during Quaternary time. Rates of subsidence measured by Woodward-Clyde Federal Services (1996) are 1 to 3 ft over 1,000 years. Radiocarbon dating of wood fragments collected from 116.5-ft below land surface in Colorado alluvium during the DOE 2002 field investigation at location MOA-435 indicates an age of 45,340 years (Appendix D, Calculation X0020900). This translates to an average subsidence rate of 2 ft per 1,000 years. This deformation is expected to occur as a slow process over time and would result in negligible probability for collapse of the tailings pile.

5.1.2.4 Geologic Resources

In the site area, potash- and magnesium-bearing sylvite and carnallite are probably present in the salt wall, estimated to be at least 9,000 ft high and composed of the Paradox Formation in Moab Valley and adjacent Spanish Valley (Doelling et al. 2002). Similar deposits underlie Cane Creek anticline about 8 miles southeast of the site that have been commercially extracted by dissolution mining. Information is not sufficient to assess the extractability or value of the saline deposits and associated economic elements in the site area and Moab Valley.

Brine has also been produced from salt beds in the Paradox Formation underlying Moab Valley about 2.5 miles southeast of the site. There, fresh water was injected in a well, and brine containing about 310,000 mg/L (31 percent) sodium chloride was recovered in a nearby well. Brine production forms large caverns in the salt beds (Mayhew and Heylmun 1965) that have been used to store liquefied petroleum gas. The salt is currently being used for this purpose about 1.5 miles southeast of the site.

No oil or gas resources are known at the site. Two oil and gas test holes were drilled within 1 mile of the site, and several other test holes were drilled in Moab-Spanish Valley less than 5 miles from the site; all were unsuccessful and abandoned. Near the site, the first test hole drilled by Embar Oil Company in 1926, just 0.5 mile northeast of the site, was abandoned at a depth of only 300 ft. The second test hole near the site was drilled from 1920 to 1928 by Embar Oil-Big Six Oil Companies less than 0.5 mile south of the site; several shows of oil and gas occurred, but the hole was abandoned at a depth of 5,345 ft in the Paradox or Honaker Trail Formation. Some oil and gas production occurs in the Paradox Formation elsewhere in this region—the closest is the Bartlett Flat field about 10 miles west-southwest of the site.

The modern and older alluvium along the Colorado River, covering much of the site outside the tailings pile, contains sand and gravel suitable for highway and other construction. The considerable thickness of alluvial basin fill (up to 500 ft) in Moab Valley beneath the site may also contain significant sand and gravel resources. A sand and gravel pit adjacent to the west edge of the site near the junction of US-191 and SR-279 was used by the Utah Department of Transportation for highway construction and maintenance. The present status of this pit, UDOT 19076 (McDonald 1999), in older Colorado River alluvium and Moab Wash detritus appears to be inactive.

Uranium and vanadium prospects in the form of several short adits occur just south of the site, south of SR-279 along the lower slopes of Poison Spider Mesa. These workings are in arkosic sandstone beds in the Honaker Trail Formation and in the lower part of the Chinle Formation. Doelling et al. (2002) reported that the radiation levels in the Honaker Trail workings were about twice background level, and those of the lower Chinle were about background level. No significant uranium-vanadium deposits are known from the Honaker Trail Formation in this region; however, uranium and copper have been produced from sandstone and conglomeratic beds containing carbonaceous debris in the lowermost part of the Chinle Formation in the Seven Mile Canyon area about 8 miles northwest of the site. Carbonaceous debris necessary for the formation of uranium (and copper) deposits in the lower Chinle Formation is not present in the site and nearby area, and radiation levels are low, both in the Honaker Trail and Chinle Formations; therefore, it is unlikely that uranium-vanadium (or copper) deposits of economic significance are present in the site area.

5.2 Hydrology

5.2.1 Surface Water

The Colorado River, located along the east boundary of the site, is the only significant surface water feature associated with the Moab site (Figure 1−2). The following sections describe the river hydrology, historical flows, nearby tributaries, river surface elevation, and gradients.

5.2.1.1 River Hydrology

The river enters Moab Valley northeast of the site through a canyon formed by outcrops of the Wingate and Kayenta Formations. After entering the valley, the river flows approximately 3 river miles across the valley to the south prior to exiting the valley through a narrow slot in the bedrock cliffs locally known as the Portal. The Portal is a geologic feature dominated by Cutler-Rico, Moenkopi, and Chinle Formations. Along the 3 miles through the valley, the river is bounded to the north by the coalesced alluvial fans of Moab and Courthouse Washes, and to the south by a large topographic depression known as the Matheson Wetlands Preserve.

Mussetter and Harvey (1994) identified the Moab Wash-Courthouse Wash alluvial fan complex, the talus-mantled slopes on the bank located downstream of the tailings pile, and the Matheson Wetlands Preserve as the three principal geomorphic features controlling bank erosion potential within the valley. Because both the river entrance to the north and east and exit to the south are bedrock-controlled, the potential for lateral migration of the river within the valley is severely constrained by inlet trajectory and the distance between the bedrock-fixed inlet and exit control points (Mussetter and Harvey 1994).

A sandbar is present at the mouth of Moab Wash and may extend approximately 1,300 ft downstream of the mouth, depending on the river stage. The presence of this sandbar significantly reduces river flow velocity along the western bank of the Colorado River and produces several shallow backwater areas between the sandbar and the shoreline during low river stages. These backwater areas may provide potentially suitable habitat for the Colorado pikeminnow.

5.2.1.2 Main Channel Flows

Colorado River flow data (in cubic ft per second) have been collected from the USGS Cisco, Utah, gaging station (Station No. 09180500) since 1914. The drainage area above the gage is 24,100 square miles, and the gage is located approximately 1 mile downstream of the confluence of the Colorado and Dolores Rivers. The station is also located approximately 31 river miles upstream of Moab and represents the closest gaging station to the site along the Colorado River. Only minor tributaries exist between the gage location and the site, and the average discharge of the Colorado River in the vicinity of Moab is estimated to be less than 1 percent more than the average discharge measured at the Cisco gaging station (Utah Department of Natural Resources 1971).

Cooper and Severn (1994) noted a distinct break in the frequency of flows over 40,000 cfs since 1959, at which time upstream flow regulation began to impact river flows. Calculating the average of annual mean flows supports this change in the historical Colorado River flows. From 1914 through 1958 (excluding the years 1918 through 1922, when Colorado River flow data at the Cisco gaging station were not recorded), the average daily mean flow was approximately 7,800 cfs. From 1959 through 2003, the average daily mean flow was approximately 6,780 cfs.

The average peak flow data have a similar trend. Prior to 1959, the average peak flow was approximately 43,400 cfs, while post-1959 flow data show an average peak flow of approximately 28,000 cfs. Daily mean discharge measured at the Cisco gaging station from 1959 through 2003 are presented in Figure 5–3.

The annual flow pattern in the form of the minimum, maximum, and average daily mean discharges measured at the Cisco gaging station from 1959 through 2003 is presented in Figure 5–4. Base river flow typically ranges from 3,000 to 5,000 cfs for most of the year. Between April and July the river discharge and stage dramatically increase in response to snowmelt runoff. On average, the river stage rises approximately 7 ft during peak flows at the Cisco gaging station.

Data relating river flow to changes in river stage in the vicinity of the site are limited. During the 2003 spring runoff, the daily mean peak flow of 26,200 cfs resulted in an 8.5-ft increase in the river surface elevation at the former pump house location (where the stilling well is located). Cooper and Severn (1994) measured a river stage increase of approximately 9.5 ft at a staff gauge located along the Matheson Wetlands Preserve in response to the 1993 runoff that peaked at a flow of 49,300 cfs.

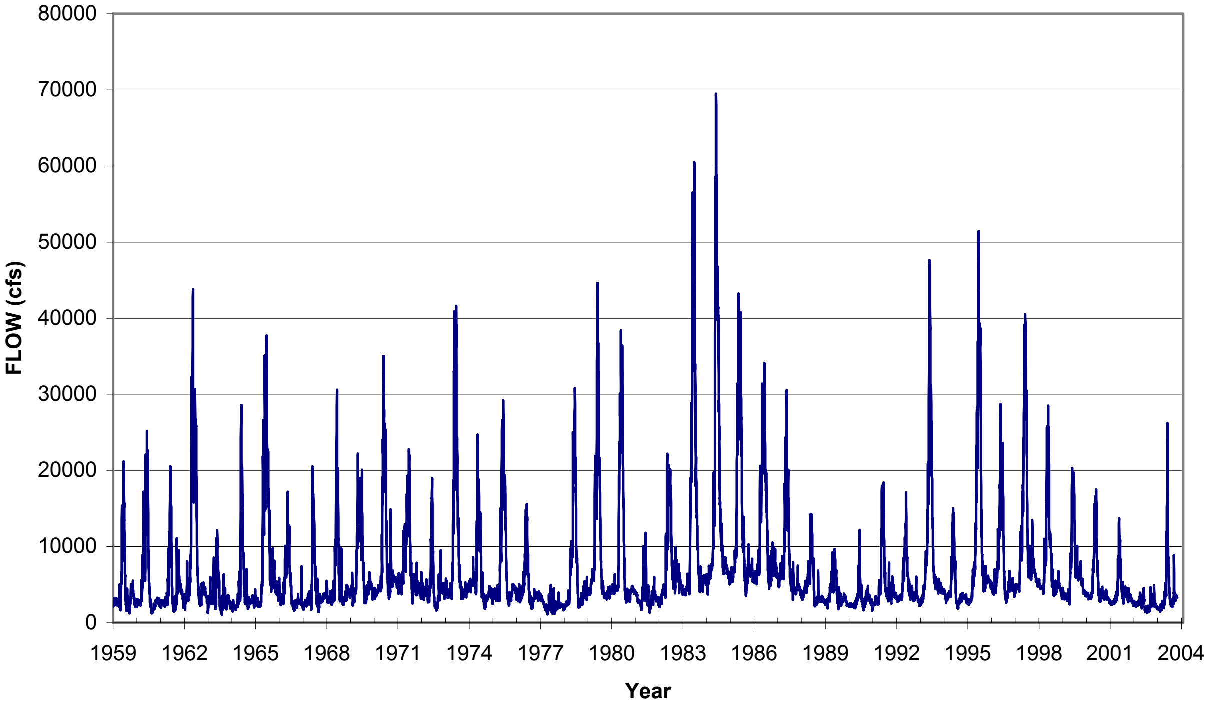


Figure 5–3. Average Peak Flows for the Colorado River Measured at the Cisco Gaging Station, 1959 through 2003

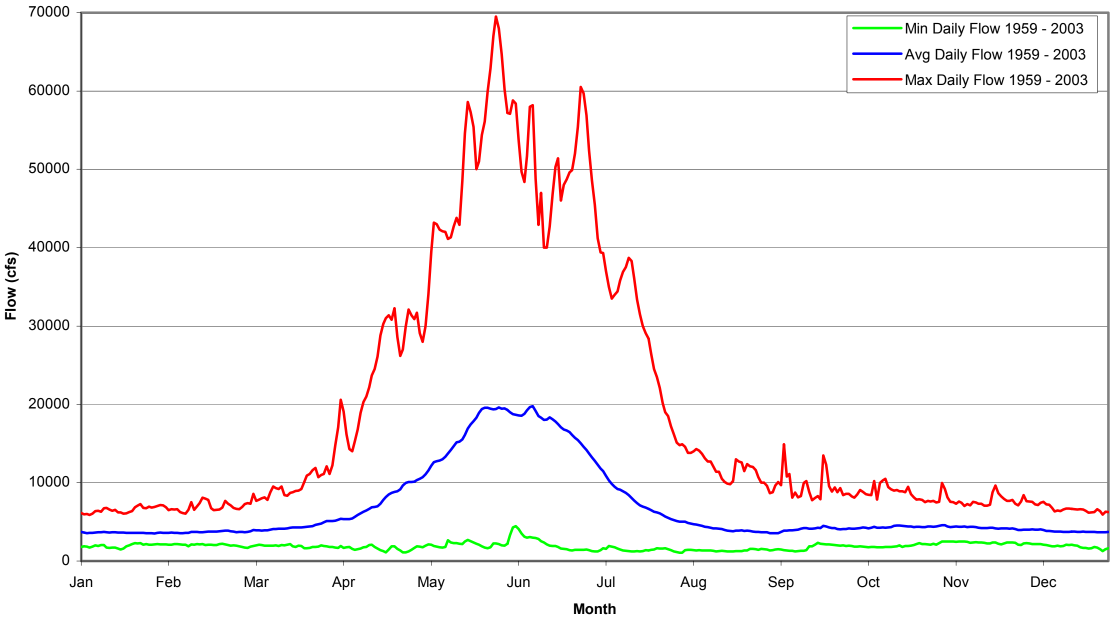


Figure 5–4. Minimum, Maximum, and Average Daily Mean Flows Measured at the Cisco Gaging Station, 1959–2003

Mussetter and Harvey (1994) identified two Colorado River flows that are significant for the Moab site. At a flow of approximately 40,000 cfs, the river elevation exceeds its bank and floods the Matheson Wetlands Preserve. The years and corresponding number of days during each year that the flow has exceeded 40,000 cfs are presented in Table 5–1.

Table 5–1. Peak Flows and Corresponding Number of Days With Flow Greater Than 40,000 cfs, 1959–2002

Year	Peak Flow (cfs)	Number of Days Exceeding 40,000 cfs
1962	44,400	4
1973	42,800	3
1979	45,400	3
1983	61,900	27
1984	70,300	46
1985	43,900	9
1993	49,300	16
1997	41,200	2

Note: Flows Measured at the Cisco Gaging Station

The other critical flow occurs at about 70,000 cfs, which, according to Mussetter and Harvey (1994), produces a river elevation such that river water comes in contact with the toe of the tailings pile. Analysis of the flow data from the Cisco gaging station indicates there has been only one day (in 1984) since 1959 in which the flow has exceeded 70,000 cfs.

5.2.1.3 Magnitude of Ground Water Response to Changes in River Elevation

Historical ground water elevations in several wells and corresponding Colorado River flows were reviewed to assess the temporal response of the aquifer to changes in river flows and stages. The river discharge data used for this analysis came from the Cisco gaging station. Ground water data were drawn partly from databases for the ATP series wells, for which mostly monthly water levels have been recorded since 1982. Additional water level data were retrieved for the AMM series wells, which were installed in 1989. The locations of the ATP and AMM wells are shown in Plate 1. Well completion data for the ATP and AMM wells are summarized in Table 5–2.

Graphs of ground water elevation (ft msl) fluctuations at ATP and AMM wells in response to Colorado River flows (cfs) are presented in Appendix B. The data used to construct these graphs do not comprise complete information sets because water levels were only measured periodically by hand and not continuously using transducers. Consequently, some minor ground water responses to changes in river flow might not be evident.

Table 5–2. Well Completion Data for the ATP and AMM Series Wells

Well	Approx. Perpendicular Distance from Present River Location (ft)	Approximate Ground Surface Elevation (ft msl)	Approximate Top of Screen Elevation (ft msl)	Approximate Bottom of Screen Elevation (ft msl)
ATP-1-S	500	3968	3823	3813
ATP-1-IS	500	3968	3753	3748
ATP-1-ID	500	3968	3673	3668
ATP-1-D	500	3968	3573	3568
ATP-2-S	700	3964	3935	3925
ATP-2-D	700	3964	3884	3869
ATP-3	3100	3997	3944	3934
AMM-1	700	3970	3953	3913
AMM-2	225	3966	3956	3916
AMM-3	1050	3966	3936	3916

Despite the discontinuous record of ground water levels, the graphs presented in Appendix B do indicate that all ATP wells respond to peak discharges in the Colorado River. Though the magnitude of the ground water response varies depending on location, it does not appear to be strongly affected by well screen depths. Ground water levels in the ATP-1 wells, with center-of-screen depths that range from 100 to approximately 400 ft bgs, all rise roughly the same amount during each peak flow. The same is true for the two ATP-2 completions, whose screen centers occur at depths of approximately 30 and 80 ft bgs. Despite being located more than 3,000 ft from the river, the water levels measured in the single ATP-3 well also exhibit some response to the peak flows.

Graphs of monitored water levels in each of the AMM wells, which are located anywhere from 200 to greater than 1,000 ft from the river also indicate clear responses to changes in Colorado River stage.

5.2.1.4 Colorado River Gaining and Losing Conditions

Transducers and data loggers were installed in wells SMI-MW01, 0406, and SMI-PW02 for the purpose of collecting continuous water level data during 2002 and 2003. These data were used to ascertain whether the Colorado River occasionally lost flow to ground water at locations on the down-gradient edge of flow paths leading from each well to the river. To make this assessment, it was necessary to estimate river surface elevations at each of these locations. This was accomplished by extrapolating measured river elevations at the stilling well located at the former mill pump house to the various river locations (all of which are downstream of the pump house) using an average water surface drop measured during four river surveys (Section 4.4.1). Well completion data for the three wells used in this analysis are presented in Table 5–3.

Table 5–3. Well Completion Data for SMI-MW01, 0406, SMI-PW02, and SMI-PZ3S

Well	**Approx. Perpendicular Distance from Present River Location (ft)**	**Approximate Ground Surface Elevation (ft msl)**	**Approximate Top of Screen Elevation (ft msl)**	**Approximate Bottom of Screen Elevation (ft msl)**
SMI-MW01	125	3967	3952	3937
0406	150	3968	3955	3950
SMI-PW02	300	3966	3946	3906
SMI-PZ3S	850	3973	3951	3946

Graphs showing measured ground water levels at wells SMI-MW01, 0406, and SMI-PW02 and comparable river elevations during 2002-2003 are presented in Figure 5–5 through Figure 5–7, respectively. These results suggest that, during nearly all of 2002, ground water discharged to the river. The exception to this rule occurred during a few very short periods in early fall of the year. This finding was not surprising given that river flows in 2002 were some of the lowest ever measured at the Cisco gaging station (Figure 5.4) during 90 years of data collection. The maximum river discharge at the gaging station during 2002 was limited to approximately 4,500 cfs.

Comparison of river and ground water elevations between February and June of 2003 also indicates that ground water was discharging to the Colorado River during most of the year. However, in contrast to 2002, a period of consistently high river flows during late May and early June caused a reversal in flow between the alluvial aquifer and the river, with the river losing water to the aquifer continuously for tens of days. This losing river condition appeared to occur when the river surface reached an elevation of about 3954.5 ft msl downgradient of SMI-PW02, and a river elevation of about 3955 ft msl downgradient of wells SMI-MW01 and 0406. Several days after peak river elevations were reached, ground water discharge to the river resumed; the river elevations at which this resumption of background conditions occurred were generally 1.5 to 2 ft higher than the river elevations at which losing conditions were first observed.

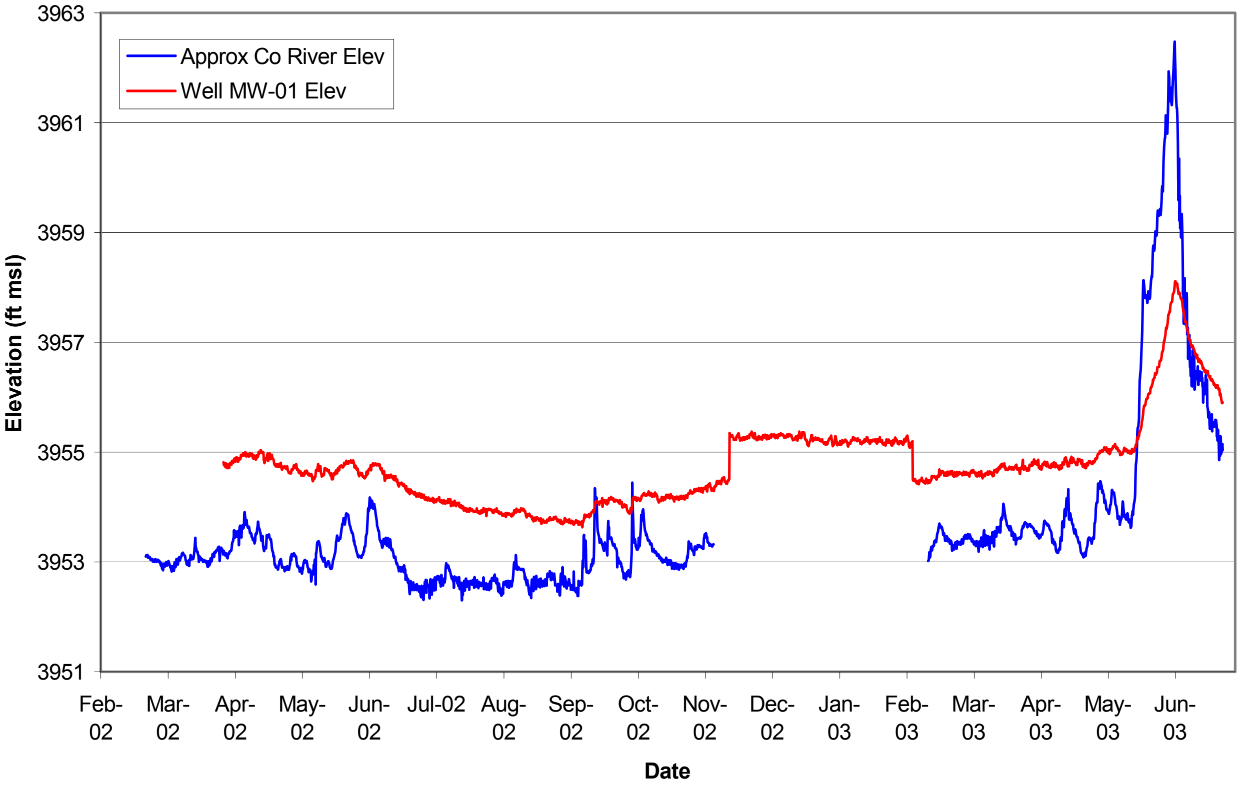


Figure 5–5. Well SMI-MW01 Ground Water Elevations and Comparable Colorado River Elevations in 2002 and 2003

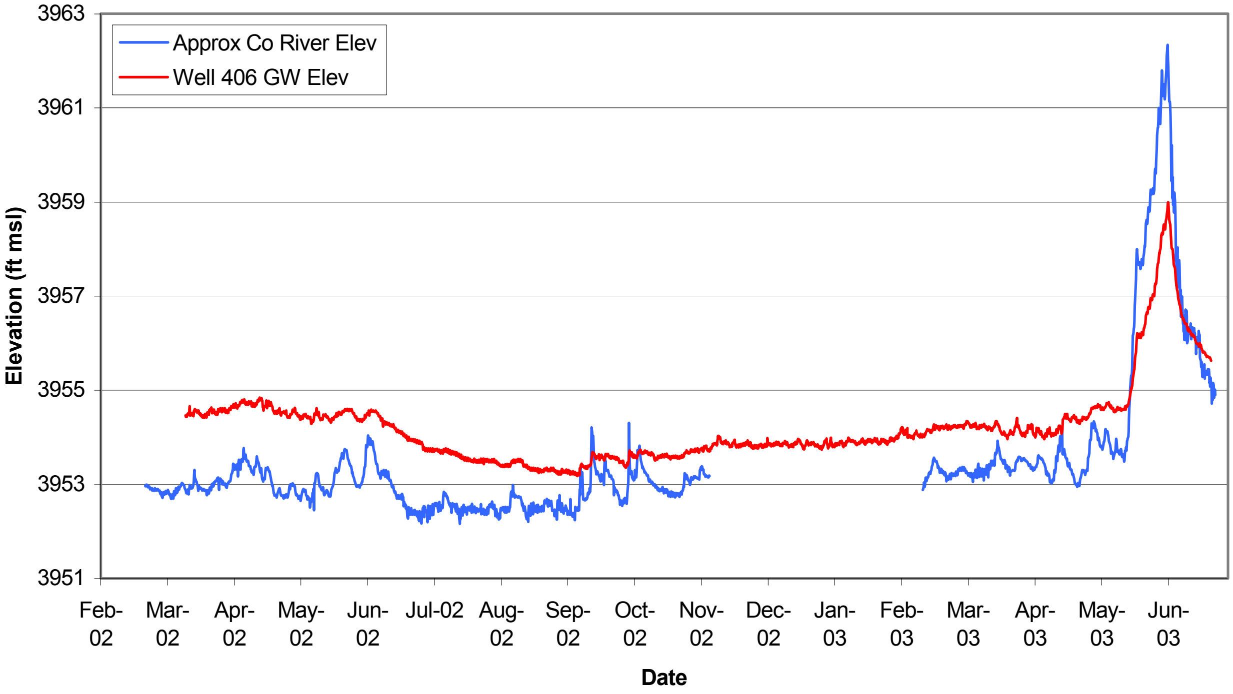


Figure 5–6. Well 0406 Ground Water Elevations and Comparable Colorado River Elevations in 2002 and 2003

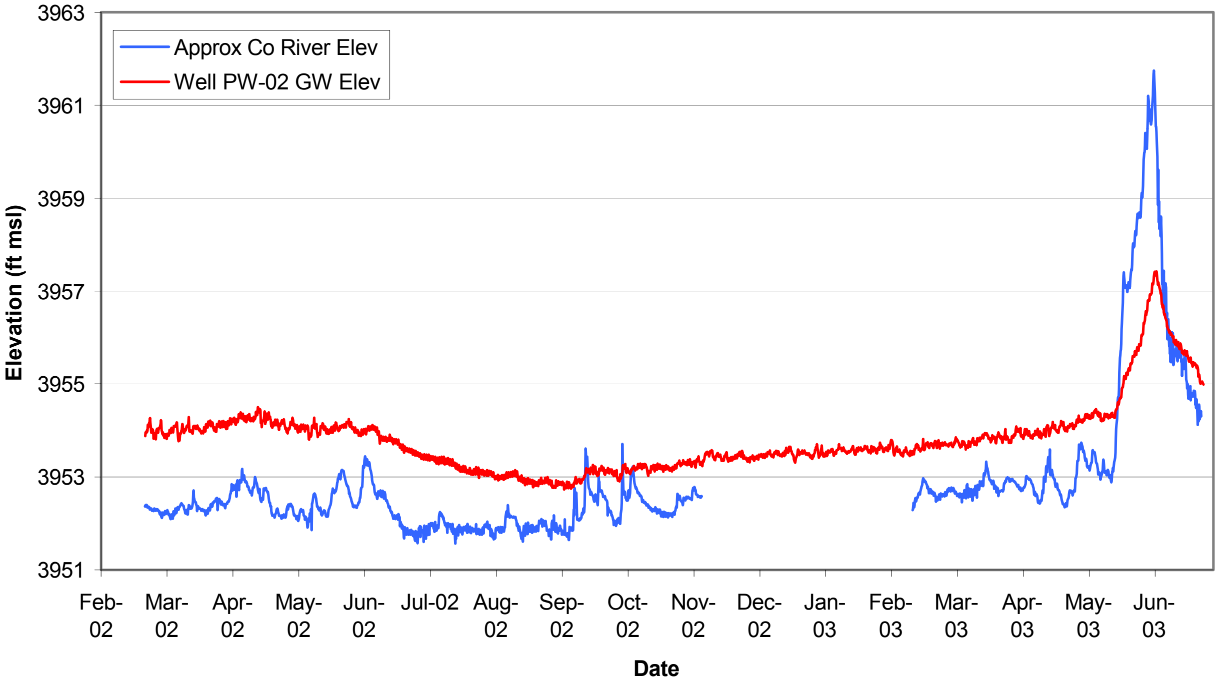


Figure 5–7. Well SMI-PW02 Ground Water Elevations and Comparable Colorado River Elevations in 2002 and 2003

5.2.1.5 Lag Time Between Peak River Flow and Peak Ground Water Elevation

Measured water levels in two wells—SMI-MW01 and SMI-PZ3S—were used to estimate the lag time between changes in Colorado River stage and associated changes in ground water level. These wells are located approximately on the same ground water flow path extending from north of the tailings to the river. Well completion data for SMI-MW01 and SMI-PZ3S, which are located about 125 and 850 ft upgradient of the river, respectively, are presented in Table 5–3.

Figure 5–8 presents measured water levels in the two monitoring wells along with the estimated river surface elevation at the end of the ground water flow path passing through the wells during 2002 and 2003. As expected, the well located closest to the river (SMI-MW01) shows a quicker response to changes in river elevation than do water levels in the farthest well (SMI-PZ3S). Though it is not clear from Figure 5–8 the data used to prepare this graph indicate that it takes from 24 to 48 hours for the ground water system approximately 125 ft away to respond to Colorado River peak flows. It takes from 72 to 96 hours for the ground water system 850 ft away to respond to the same peaks.

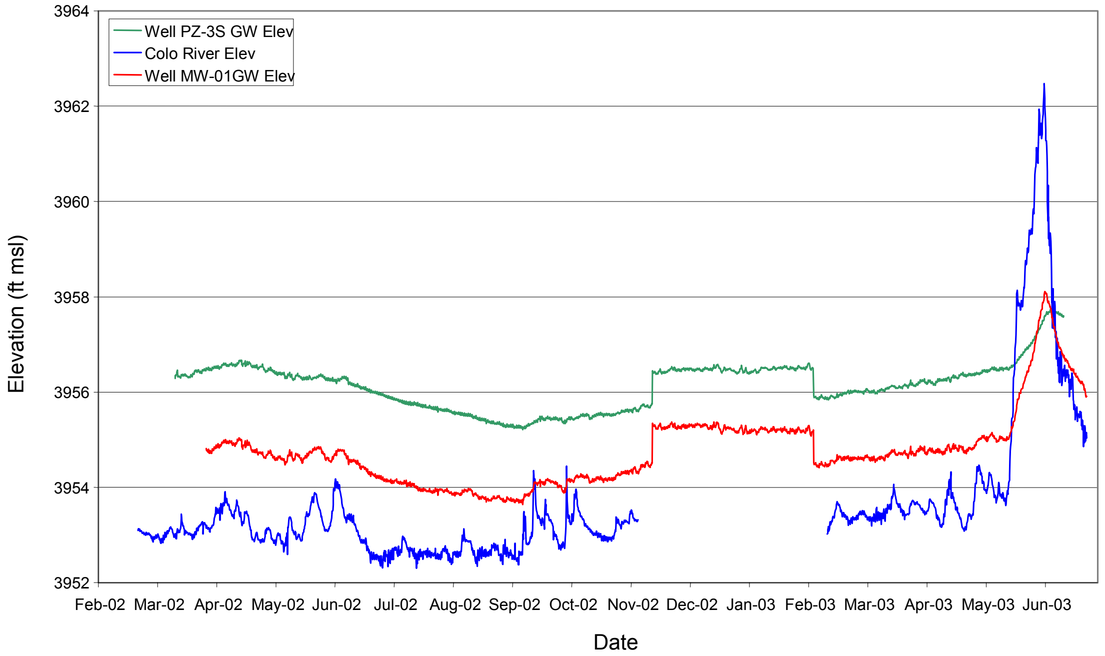


Figure 5–8. Wells SMI-MW01 and SMI-PZ3S Ground Water Elevations and Comparable Colorado River Elevations in 2002 and 2003

5.2.1.6 Tributaries

Moab Wash and Courthouse Wash are minor tributaries that provide water to the Colorado River in the vicinity of the site. Moab Wash is an ephemeral stream that crosses the site's northwest corner, and Courthouse Wash (also an ephemeral stream) discharges into the Colorado River approximately 0.5 mile upstream of the tailings pile. There are no flow data associated with Moab Wash, which was rerouted east of the mill during operations to mitigate flooding potential during peak flows.

A USGS gaging station measured flows from 1950 through 1988 on Courthouse Wash at two locations (Station numbers 09182900 and 09183000). The drainage area for Courthouse Wash is 162 square miles. Historical flow data indicate that annually the wash contained less than 1 cfs most of the time. Peak flows are in response to intense precipitation events during the late summer months. The average daily mean flow in Courthouse Wash from 1950 through 1988 (with the exception of 1954 through 1958, and 1966, when flow data were not recorded) during days of flow was 1.8 cfs. Over the same time period, the average annual peak flow was approximately 2,950 cfs.

5.2.2 Ground Water

Ground water aquifers in the Moab region occur in the unconsolidated Quaternary material deposited on the floor of Moab and Spanish Valleys and in consolidated bedrock formations. Rush et al. (1982), Weir, Maxfield, and Hart (1983), and Weir, Maxfield, and Zimmerman (1983) grouped the aquifers into a lower and upper hydrologic system. The upper ground water system consists of the unconsolidated and bedrock formations above the very low permeability salt beds of the Paradox Formation. The lower ground water system includes all stratigraphic units below the Paradox Formation. The salt beds of the Paradox Formation confine units in the regional lower system and occur over most of Moab and Spanish valleys. The Paradox Formation also underlies the Moab site. Descriptions of major stratigraphic units above the Paradox Formation that have bearing on their capacities to transmit and store water are presented in Figure 5–9.

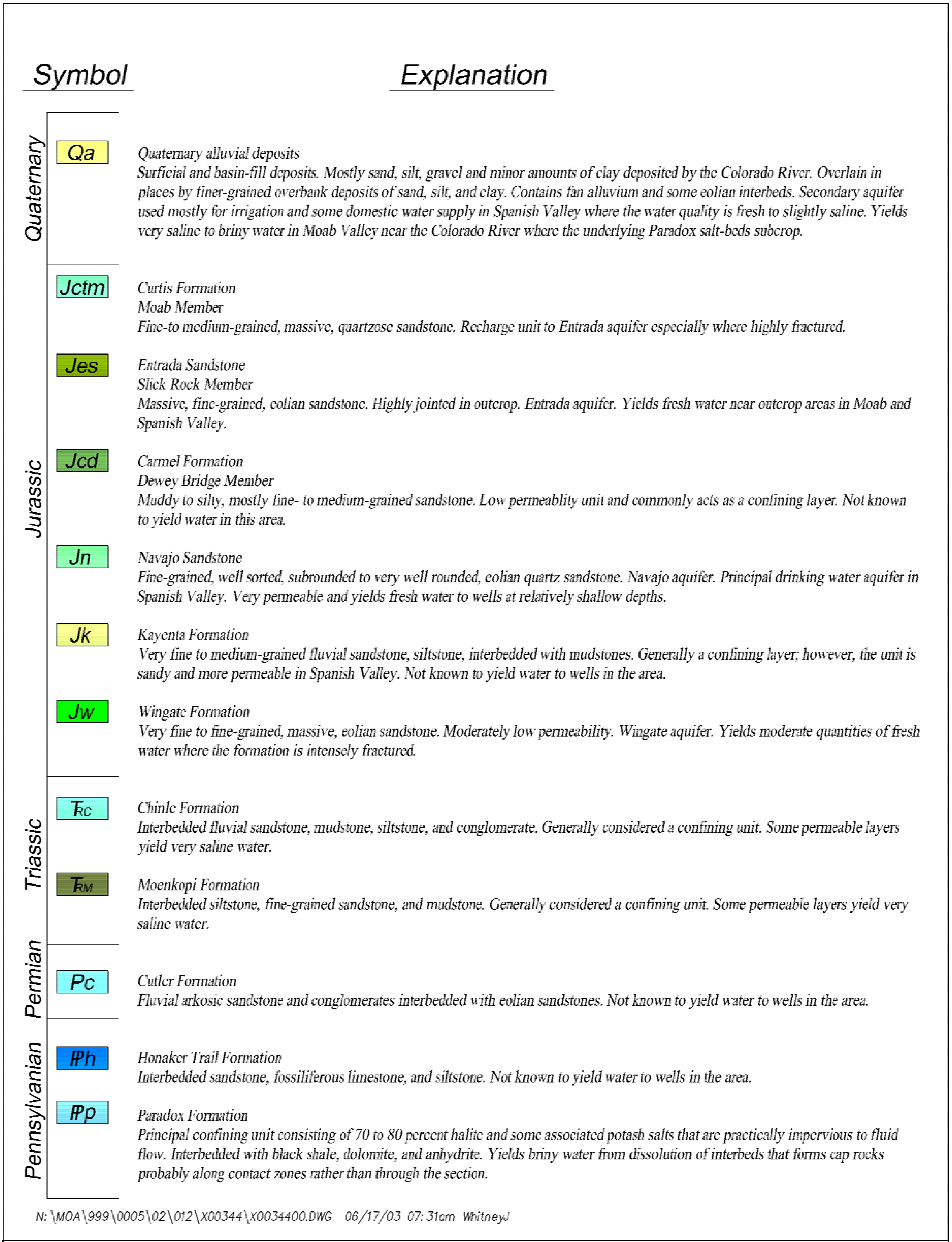
Site-related ground water contamination occurring in unconsolidated alluvium in the upper hydrologic system is the primary focus of ground water investigation at the Moab site. For ease of discussion, the ground water hydrology of the area is presented with regard to the properties of three general types of hydrogeologic units present at the Moab site: the unconsolidated alluvial aquifer, bedrock aquifers, and bedrock aquitards.

5.2.2.1 Alluvial Aquifer

Unconsolidated alluvial deposits in the region make up a secondary aquifer used mostly for irrigation and some domestic water supply in Spanish Valley (Steiger and Susong 1997). Average saturated thickness of the mostly gravelly sand alluvium is approximately 70 ft (Sumsion 1971); the greatest saturated thickness may exceed 450 ft in Moab Valley near the Colorado River (Doelling et al. 2002). More than 200 wells are completed in unconsolidated sediments in the Moab-Spanish Valley area (Sumsion 1971) and range in depth from 30 to 300 ft (Eisinger and Lowe 1999).

The unconsolidated alluvial aquifer at the Moab site is contained mostly within two distinct depositional facies: the Moab Wash alluvium and the basin-fill alluvium. Moab Wash alluvium is composed of fine-grained sand, gravelly sand, and detrital material that has traveled down Moab Wash and grades and interfingers near the northwest boundary of the site into the basin-fill alluvium deposited by the ancestral Colorado River.

Two distinct types of material are observed in the basin-fill alluvium. The upper type consists mostly of a fine-grained alluvium (fine sand, silt, and clay), which ranges in thickness from 15-ft near the river to 40-ft in the northern and northwestern portions of the site and extends into the saturated zone in some areas. This shallow unit, referred to as the silty-sand unit, probably represents mostly fine-grained overbank deposits from the Colorado River. The lower part of the basin-fill alluvium consists mostly of a gravelly sand and sandy gravel, with minor amounts of silt and clay. The gravel clasts typically consist of subrounded pebbles and cobbles of resistant crystalline rocks that have been eroded and transported from metamorphic and igneous terranes present in the upper Colorado River Drainage Basin. This coarser alluvium, referred to as the gravelly unit, thins and pinches out to the northwest along the subsurface bedrock contact and thickens to the southeast toward the river to over 450 ft near the deepest part of the basin. Most of the borings drilled within the site boundary penetrate both the upper silty-sand unit and the lower basin-fill gravelly unit. Plate 10 presents a fence diagram showing the thickness of each unit across the site.



	Symbol	Explanation
Quaternary	Qa	Quaternary alluvial deposits Surficial and basin-fill deposits. Mostly sand, silt, gravel and minor amounts of clay deposited by the Colorado River. Overlain in places by finer-grained overbank deposits of sand, silt, and clay. Contains fan alluvium and some eolian interbeds. Secondary aquifer used mostly for irrigation and some domestic water supply in Spanish Valley where the water quality is fresh to slightly saline. Yields very saline to briny water in Moab Valley near the Colorado River where the underlying Paradox salt-beds subcrop.
Jurassic	Jctm	Curtis Formation Moab Member Fine-to medium-grained, massive, quartzose sandstone. Recharge unit to Entrada aquifer especially where highly fractured.
	Jes	Entrada Sandstone Slick Rock Member Massive, fine-grained, eolian sandstone. Highly jointed in outcrop. Entrada aquifer. Yields fresh water near outcrop areas in Moab and Spanish Valley.
	Jcd	Carmel Formation Dewey Bridge Member Muddy to silty, mostly fine- to medium-grained sandstone. Low permeablity unit and commonly acts as a confining layer. Not known to yield water in this area.
	Jn	Navajo Sandstone Fine-grained, well sorted, subrounded to very well rounded, eolian quartz sandstone. Navajo aquifer. Principal drinking water aquifer in Spanish Valley. Very permeable and yields fresh water to wells at relatively shallow depths.
	Jk	Kayenta Formation Very fine to medium-grained fluvial sandstone, siltstone, interbedded with mudstones. Generally a confining layer; however, the unit is sandy and more permeable in Spanish Valley. Not known to yield water to wells in the area.
	Jw	Wingate Formation Very fine to fine-grained, massive, eolian sandstone. Moderately low permeability. Wingate aquifer. Yields moderate quantities of fresh water where the formation is intensely fractured.
Triassic	TRc	Chinle Formation Interbedded fluvial sandstone, mudstone, siltstone, and conglomerate. Generally considered a confining unit. Some permeable layers yield very saline water.
	TRm	Moenkopi Formation Interbedded siltstone, fine-grained sandstone, and mudstone. Generally considered a confining unit. Some permeable layers yield very saline water.
Permian	Pc	Cutler Formation Fluvial arkosic sandstone and conglomerates interbedded with eolian sandstones. Not known to yield water to wells in the area.
Pennsylvanian	IPh	Honaker Trail Formation Interbedded sandstone, fossiliferous limestone, and siltstone. Not known to yield water to wells in the area.
	IPp	Paradox Formation Principal confining unit consisting of 70 to 80 percent halite and some associated potash salts that are practically impervious to fluid flow. Interbedded with black shale, dolomite, and anhydrite. Yields briny water from dissolution of interbeds that forms cap rocks probably along contact zones rather than through the section.

After Doelling and others, 2002; Blanchard 1990; Weir and others, 1983; Rush and others, 1982; Sumsion 1971

Figure 5–9. Regional Hydrogeologic Characteristics of Major Stratigraphic Units

The silty-sand unit is generally less permeable than the underlying gravelly unit. Measured hydraulic conductivity (K) values for the basin fill alluvium are summarized in Table 3–12 and Table 4−16. Hydraulic conductivity of the basin-fill alluvium has a wide range of values. The hydraulic conductivity in the silty-sand unit is typically less than 2 ft/day (Table 3−12); estimates of this parameter are based on laboratory tests of sediment samples and field testing at depths of up to 13 ft in Borehole 8 (Dames and Moore 1975). Aquifer pumping tests performed in the gravelly unit indicate that its hydraulic conductivity ranges from 93 to 202 ft/day (SMI 2001) and averages 154 ft/day. SMI also performed slug tests in gravelly basin-fill deposits, which produced hydraulic conductivity estimates that ranged from 2.1 to 80 ft/day and averaged 41 ft/day. The range of estimates obtained previously by SMI (Section 3) and DOE (Section 4) indicate that K values of 100 to 150 ft/day are generally representative of the gravelly unit of the basin-fill alluvium.

Ground water contamination would be expected to migrate more slowly in the upper silty-sand unit than in the lower gravelly unit. Because electrostatic forces tend to be more prevalent in clayey materials, and soil particle surface areas are typically larger in fine-grained alluvium, the silty-sand unit is also expected to geochemically retard the migration of some inorganic contaminants more than the gravelly unit. Because of its higher hydraulic conductivities, the lower gravelly unit might also provide a preferential flow path for contaminant migration.

5.2.2.2 Bedrock Aquifers

The Entrada Sandstone is capable of transmitting and yielding small quantities of water but is not important as a water resource (Sumsion 1971). In the past, the Entrada was described as consisting of three geologic units: the Moab Member (youngest), the Slick Rock Member, and the Dewey Bridge Member (oldest) (Wright et al. 1962; Doelling 1985; Peterson 1988; Doelling and Ross 1998). This report follows the more recent stratigraphic classification proposed by O’Sullivan (2000) and Doelling et al. (2002) that assigns the Dewey Bridge Member to the Carmel Formation, the Slick Rock Member to the Entrada Sandstone, and the Moab Member to the Curtis Formation, which overlies the Entrada. The Carmel Formation thickness ranges from 90 to 110 ft in the study area. This red-brown, muddy to silty, mostly fine- to medium-grained sandstone is less permeable than the Entrada Sandstone and commonly acts as a confining layer. As a consequence, the Entrada Sandstone and Curtis Formation tend to act more like aquifers in the region than does the Carmel Formation.

The Entrada Sandstone yields relatively fresh water and is referred to as the Entrada aquifer. It is a massive eolian sandstone estimated to be a least 250 ft thick, is very fine to fine-grained, well indurated, and highly fractured near the Moab Fault zone (Doelling et al. 2002). The Entrada aquifer subcrops beneath the alluvial aquifer at the northwestern boundary of the site along the Moab Fault zone (Plate 11). The younger Curtis Formation ranges from 60 to 100 ft in thickness (Doelling et al. 2002). The Curtis is a fine- to medium-grained, massive quartzose sandstone that is typically highly jointed in outcrop and provides recharge to the Entrada aquifer, especially where fractured (Blanchard 1990).

The principal bedrock aquifer in the region is the Glen Canyon aquifer, which comprises three geologic units: Navajo Sandstone (youngest), Kayenta Formation, and Wingate Sandstone (oldest). The Navajo Sandstone, which ranges in thickness from 300 to 700 ft (Doelling et al. 2002), is the shallowest and most permeable formation in the Glen Canyon Group. Consequently, it is the primary target for most bedrock wells drilled in the area (Eisinger and Lowe 1999) and the principal source of drinking water in Spanish Valley (Steiger and

Susong 1997). The Wingate Sandstone is the deepest unit of the Glen Canyon Group and is about 250 to 400 ft thick in the Arches area (Doelling et al. 2002). The Wingate is a relatively homogenous unit consisting of massive cross-bedded, very fine to fine-grained eolian sandstone, which reduces the intrinsic permeability of the aquifer. However, the Wingate yields moderate quantities of water to springs where the formation is intensely fractured (Sumsion 1971). Regionally, the Kayenta Formation tends to be less permeable than the overlying Navajo Sandstone and the underlying Wingate Sandstone (Sumsion 1971) and is generally considered a confining layer. However, as noted by Blanchard (1990), the Kayenta Formation in the Spanish Valley area is sandy and is locally capable of transmitting water between the Navajo and Wingate sandstones.

The Glen Canyon aquifer receives recharge from precipitation and runoff. Direct recharge to the aquifer occurs where precipitation and runoff contact exposed bedrock. Landforms associated with the Glen Canyon include knolls and depressions that contain loose sand. The loose sand provides indirect recharge to the bedrock aquifer if it collects runoff and is thick enough to protect the collected water from evaporation. Recharge areas for the Glen Canyon Group are somewhat localized and depend on topography; however, recharge can occur anywhere over the extensive outcrop area of the bedrock aquifer. Recharge rates would be enhanced if the bedrock is fractured or jointed. The Glen Canyon aquifer can also be recharged by vertical leakage from the overlying Entrada Sandstone where it is present.

Regional discharge from the Glen Canyon aquifer within the Spanish Valley area occurs along valley margins where the aquifer contacts the basin-fill alluvium. Discharge is sometimes evident as springs and seeps, vegetation growing near the base of Glen Canyon Formations along the Colorado River, and salt encrustation and staining of rocks where occasional discharge issues from the bedrock.

Jobin (1962) originally described hydraulic conductivity of the Navajo Sandstone as ranging from less than 0.4 ft/day to 1 ft/day and transmissivity as ranging from a maximum of 700 ft^2/day to nearly zero where the formation pinches out. Jobin (1962) has estimated the hydraulic conductivity of the Wingate Sandstone to range from 0.1 ft/day to 0.4 ft/day and has estimated transmissivity to range from 40 to 150 ft^2/day. Eisinger and Lowe (1999) report that hydraulic conductivities of competent (nonfractured) sections of the Wingate and Navajo Sandstones to range from 0.1 to 0.4 ft /day and 0.0037 to 5.1 ft/day, respectively. Those authors report that fractured sections of Navajo Sandstone have hydraulic conductivities of up to 88 ft/day.

The basin-fill aquifer directly overlies a subcropped portion of the Glen Canyon aquifer (Plate 11) at the northwest boundary of the site near the Moab Fault zone. This contact was identified during the drilling of paired wells MOA-433 and -455, which DOE installed during the 2002 field investigation (Section 4.0). Collections based on water elevations measured on December 2, 2002, in these wells indicate that the estimated vertical hydraulic gradient between bedrock and basin fill at this location is -3.6×10^{-3} (a dimensionless number, negative value indicates upward gradient). Assuming a vertical hydraulic conductivity for the Glen Canyon aquifer of 0.5 ft/day, the upward component of specific discharge (flow per unit area) here is approximately 1.8×10^{-3} ft/day. However, depending upon the local characteristics of the bedrock, such as the degree of fracturing or the proximity to important faulting, the specific discharge from bedrock to alluvium in this portion of the Moab site could be higher than the calculated value.

5.2.2.3 Bedrock Aquitards

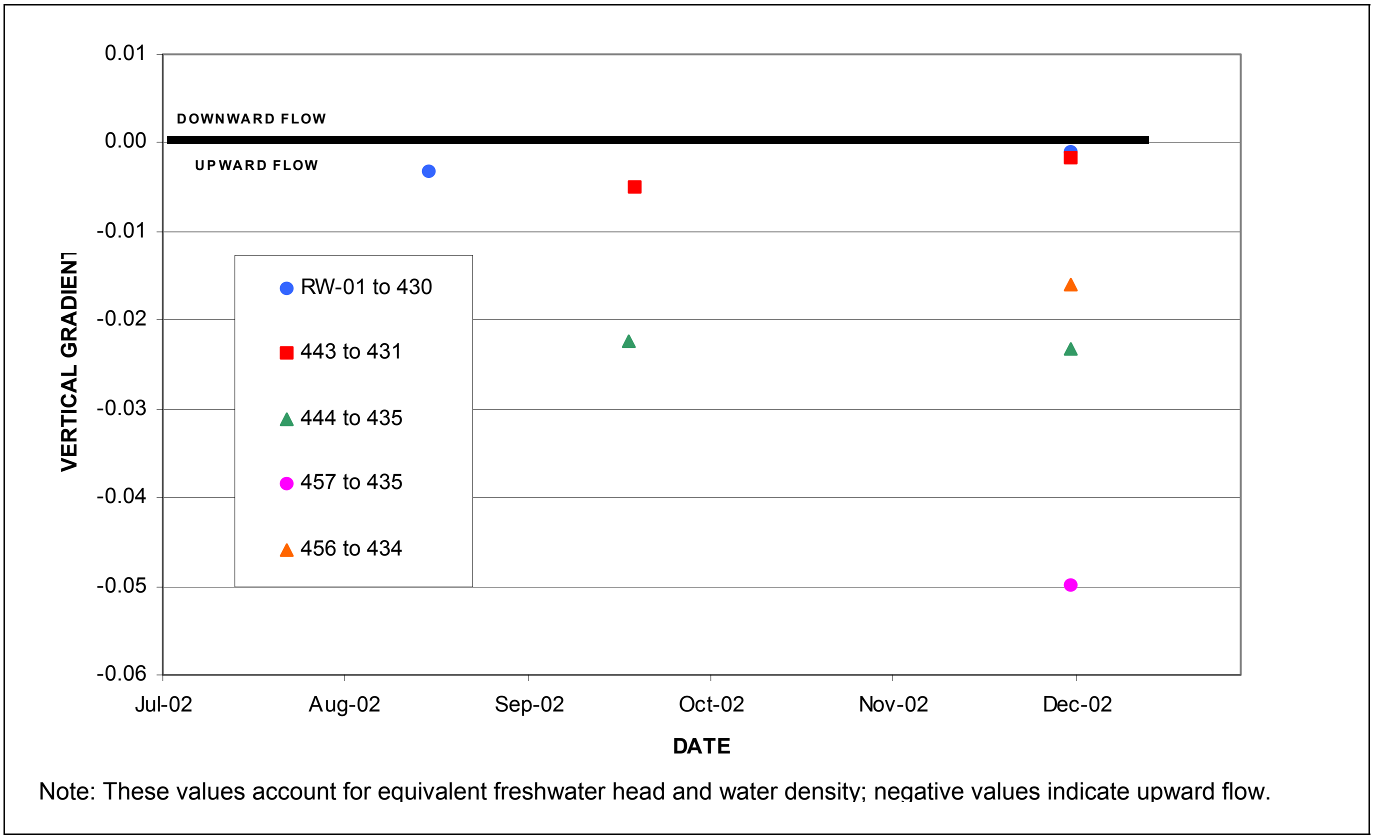
The Chinle, Moenkopi, and Paradox Formations form aquitards below the Glen Canyon aquifer (Figure 5–9). The Paradox Formation subcrops below alluvium at the site except for a small area along its northern and western boundaries, where the Chinle, Moenkopi, Cutler, and Honaker Trail Formations are in direct contact with the alluvial aquifer (Plate 11).

Dissolution of the salt within the Paradox Formation is mostly responsible for the brine within the basin fill alluvium. Water flow is believed to very minor to nonexistent within the Paradox Formation. In areas where the Chinle and Moenkopi Formations underlie the basin-fill alluvium, the potential does exist for some slight upward ground water movement from these bedrock units to the alluvium.

In areas containing brackish to briny ground water, the direction of vertical flow between adjacent hydrogeologic units depends partly on observed local TDS concentrations. In such areas, flow potential in each unit consists of two components: an equivalent freshwater head and a component defined by water density. Procedures presented in Guo and Langevin (2001, equation 32) that take into account both components allow calculation of the vertical hydraulic gradient between the Moenkopi Formation and the basin fill alluvium and the Chinle Formation and the basin fill alluvium, as presented in Figure 5–10. The magnitude of the dimensionless vertical gradient ranges from −0.001 to –0.0498 and averages −0.015. These values indicate that the direction of vertical ground water flow in either bedrock unit is upward into the basin-fill alluvium. Because both the Chinle and Moenkopi Formations contain relatively saline water, the upward specific discharge probably contributes to salinity of alluvial ground water. No tests were performed during this investigation to measure the vertical hydraulic conductivity of bedrock; however, typical ranges of reported hydraulic conductivity values for indurated sediments reported in Freeze and Cherry (1979) indicate that a value of 2.8×10^{-3} ft/day can be assumed. If this assumed hydraulic conductivity value is combined with an average vertical hydraulic gradient of −0.015 in the equation of density-dependent flow (Guo and Langevin 2001), the upward specific discharge of saline water from the Chinle and Moenkopi Formations is estimated to be 4.2×10^{-5} ft/day. This simple estimate indicates that the upward flow of water from the Chinle and Moenkopi Formations to overlying alluvium is relatively minor and suggests that it is between one and two orders of magnitude less than the upward specific discharge from the Glen Canyon aquifer.

5.2.2.4 Alluvial Freshwater-Saltwater System

The basin-fill aquifer contains natural brine (salt water) that underlies a relatively thin section of brackish to fresh water. Most of the brine probably originates through chemical dissolution of salt in the Paradox Formation. Interaction between relatively shallow southeastward-moving fresh water at the site and brine emanating from the Paradox Formation creates a flow system that is similar to those found above salt domes (e.g., Herbert et al. 1988; Oldenburg and Pruess 1995; Konikow et al. 1997). Mass transfer of dissolved salt to the upper parts of the aquifer is believed to occur both through advection and hydrodynamic dispersion, the latter of which results from both mechanical mixing processes and molecular diffusion (Freeze and Cherry 1979).



Note: These values account for equivalent freshwater head and water density; negative values indicate upward flow.

Figure 5–10. Summary of Calculated Vertical Hydraulic Gradients Between the Chinle Formation and Alluvium and the Moenkopi Formation and Alluvium

Though dissolution of salt beds in the Paradox Formation is the greatest source of brine in the alluvial aquifer, the Chinle and Moenkopi Formations probably also contribute waters of relatively high salinity to the alluvium. As discussed in Section 5.2.2.3 and illustrated in Figure 5–10, vertical hydraulic gradients that account for both freshwater head and water density effects in nested wells screened in either the Chinle or Moenkopi Formations and the overlying alluvial aquifer indicate upward flow of ground water, albeit at relatively low rates. Because TDS levels tend to be higher in the Chinle and Moenkopi Formations than in the alluvium, additional sources of saline water to the alluvial aquifer are inferred.

A conceptual model of the subsurface hydrogeology along a representative cross section showing the interface between the deeper brine and the overlying brackish to freshwater system is illustrated in Figure 5–11. The interface is assumed to exist where the TDS concentration equals 35,000 mg/L. The transition from brine to fresh water sometimes occurs over a short vertical distance; by convention, the line demarcating the boundary between brine and fresh water in such cases is typically referred to as a “sharp” interface (see Section 4.1). TDS concentrations above the interface, decrease gradually before reaching an elevation where relatively fresh water is observed. This diffuse zone is brought about by mixing of more saline water with fresher water through the process of hydrodynamic dispersion. For convenience, the term saltwater interface is used to describe the depth at which a TDS concentration of 35,000 mg/L is observed.

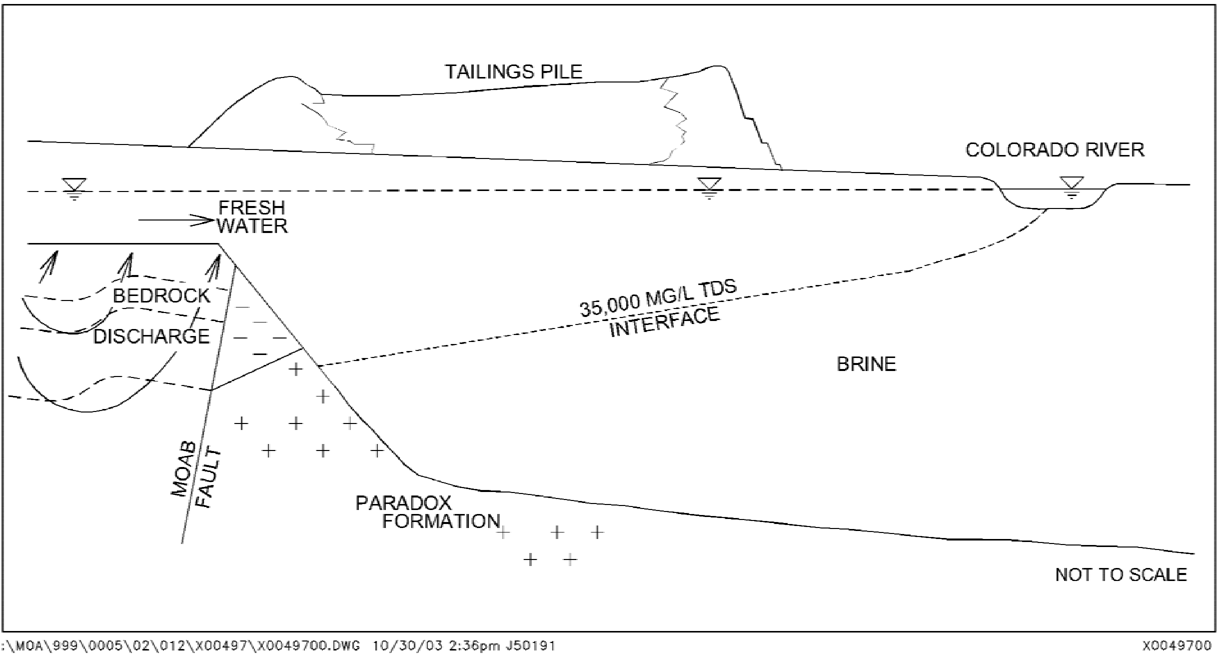


Figure 5–11. Conceptual Model of the Saltwater Interface

TDS distribution with depth below the water table in the area between the tailings and the Colorado River typically follows a distinct pattern. At shallow depths TDS concentrations in this area are typically 5,000 to 10,000 mg/L. Concentrations gradually increase with depth to levels of about 20,000 mg/L. A subsequent increase to levels of about 60,000 to 80,000 mg/L occurs over a relatively short vertical distance of about 30 to 40 ft. This increase is followed by much more gradual increases of TDS with depth. In this general pattern, the saltwater interface occurs within the zone of most rapid TDS increase with depth (see Section 4.1).

The vertical position of the saltwater interface in most natural systems is typically in a state of dynamic equilibrium because the forces that affect ground water have been present for thousands of years. Because in natural systems the fresh water flow is mostly horizontal above the interface, the fresh water can be thought of as a liquid that "floats" on salt water. At the Moab site, the interface extends across the site largely in the shape of a wedge such that the deepest part of the interface is near the northwestern edge of the site, and the shallowest part is near the Colorado River. This shape is consistent with those occurring in systems where fresh water flows over salt domes (Konikow 1997).

Three wells in the PW-01 cluster (SMI-PW01, -PZ1S, and PZ1M) were equipped with probes that measure ground water specific conductance for the purpose of gaging the response of the saltwater in the alluvial aquifer to fluctuations in Colorado River stage. Well SMI-PW01 is an ideal location for tracking the interface because its screened section encompasses the depth (58 ft btoc) at which TDS levels of 35,000 mg/L are observed in the cluster (Section 4.1). The river elevations corresponding to this cluster were estimated using a method mentioned in Section 5.2.1.4, wherein surface water levels measured at the former millsite pump house were extrapolated to downstream locations using river survey data.

Figure 5–12 presents measured specific conductances in SMI-PW01 at a depth of 58 ft btoc along with corresponding river elevations in 2002 and 2003. Also shown are measured water levels in the well during the same period. This graph indicates that, as the river stage increases (which also results in an increase of ground water elevation), the specific conductance also tends to increase. Though the cause of this response is uncertain, hydraulic phenomena associated with density-dependent flow might provide an explanation. Modeling of two-dimensional density-dependent flow and transport (presented in Appendix D, Calculation X0062600) indicates that the discharge of fresh ground water (TDS < 35,000 mg/L) to the west bank of the river occurs within a relatively small vertical band adjacent to the river. When the river elevation rises, it becomes possible for this band to rise vertically such that the thickness over which freshwater discharge occurs is the same as that observed prior to the river stage increase.

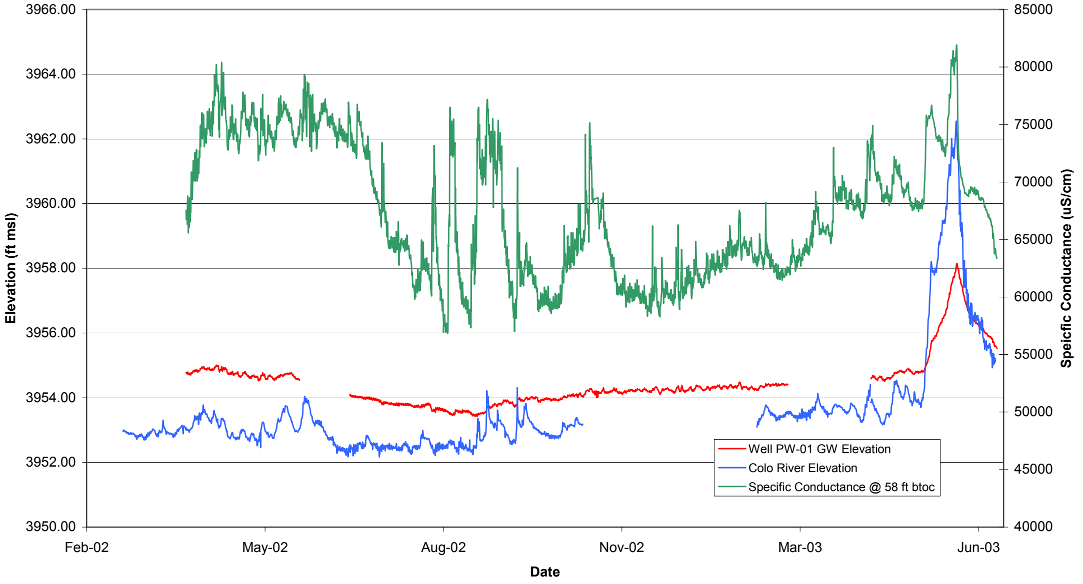


Figure 5–12. Well SMI-PW01 Specific Conductances and Ground Water Elevations and Comparable Colorado River Stages in 2002 and 2003

Specific conductance data collected from the probes in wells PZ1S and PZ1M (which are screened above and below the saltwater interface respectively) in 2002 and 2003 did not show the same response to river elevations as did the data from well SMI-PW01. This latter observation suggests that either TDS levels over the screened sections of these wells vary mildly with depth or that the well screens are located at depths that do not experience significant changes as a result of external stressors like river stage.

In a state of dynamic equilibrium, the position of the interface between the southeast toe of the tailings and the river shifts laterally and vertically in response to hydrologic stresses such as evapotranspiration from tamarisk plant communities and changes in Colorado River stage. The interface might also shift vertically upward as a result of pumping from the shallow freshwater zone (i.e., during pump-and-treat remediation); this process, referred to as upconing (Bear 1979), causes salt water to intrude zones that are normally occupied by the fresher water, and possibly degrades water locally. In addition to potentially adversely affecting the tamarisk plant communities that are providing some beneficial phytoremediation at the site, upconing just to the west of the Colorado River, if it occurred, might also bring higher ammonia and salt concentrations closer to ground surface and augment contaminant fluxes to the river. DOE (Section 4) conducted aquifer tests in the freshwater system at Moab to evaluate the response of the saltwater interface to pumping and found that the salt water can be induced to move upward (Section 4.9). The aquifer tests indicated that careful selection of well design, well location, screen depth, and pumping rates can minimize the effects of saltwater upconing. In the absence of pumping stress, the saltwater interface at the Moab site would tend to remain stable.

Contours reflective of measured water elevations in wells screened in the upper fresh to brackish water in the basin-fill alluvium are shown in Figure 5–13. West of the Colorado River, these shallow water table contours are based on average water elevations measured in 2001 and 2002. Contours east of the Colorado River in the Matheson Wetlands Preserve are based on March 2003 water elevation measurements and indicate ground water flow toward the river. The water elevation contours west of the Colorado River indicate that fresh ground water entering the site along its northwestern and northern boundaries flows southeast toward the river on top of the deeper natural brine zone. Because Figure 5–13 was constructed from ground water elevation measurements at monitor wells screened in the upper portion of the alluvial aquifer where TDS concentrations are less than 35,000 mg/L, it does not reflect data from monitor wells where brine exists near the water table at the southernmost portion of the site.

It is apparent from the ground water level contours in Figure 5–13 that shallow ground water flows southeast at the site. In general, the spacing between contours is larger in the northwest portion of the site and becomes smaller near the river. The shapes of the contours representing water levels of 3959 and 3961 ft amsl near where the Glen Canyon Group and Entrada Sandstone subcrop in the northwestern portion of the site indicate possible local inflows of fresh water. Deflection of the 3955-ft elevation contour near Courthouse Wash such that it parallels the alluvium-bedrock border also indicates a local inflow of fresh water to the alluvium. This latter potential inflow might occur through either the Wingate Sandstone or the Chinle and Moenkopi Formations, all of which subcrop below the alluvial aquifer in this area. The general gradual narrowing between ground water elevation contours indicates a steepening of the phreatic surface in the direction of the Colorado River. This steepening is partly due to a convergence of flow vertically that would occur close to a ground water discharge site (i.e., the river) even if all ground water in the alluvial aquifer were fresh. At the Moab site, vertical convergence of flow with proximity to the river is probably enhanced by the fact that the salt water interface becomes shallower as the river is approached from the west (Figure 5–11); the principles governing density-dependent ground water flow dictate that the fresh water will tend to stay above the brine, which means that the vertical interval occupied by the fresh water becomes much thinner near the river.

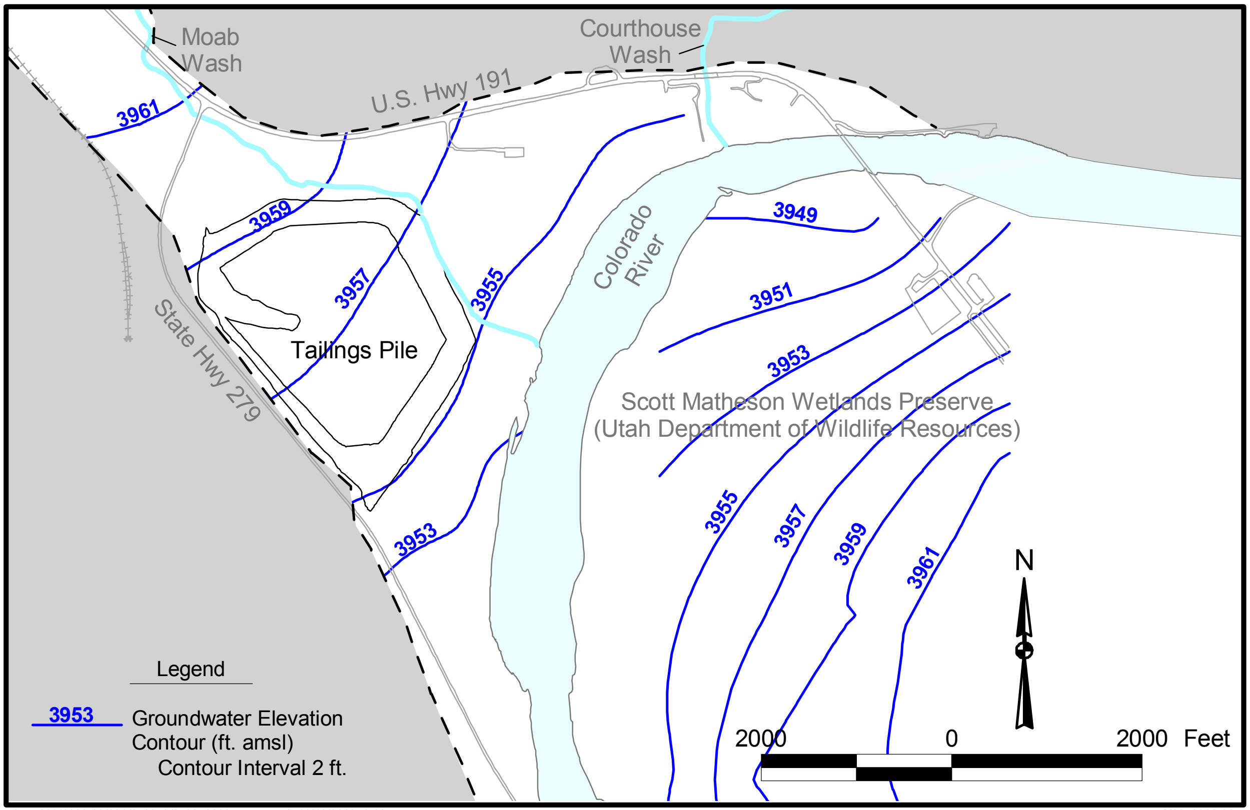


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Figure 5–13. Elevation of Upper Freshwater Surface in the Shallow Alluvium

The increasing elevation of the interface between fresh water and salt water with proximity to the Colorado River occurs largely because the river acts a discharge site for regional ground water. Near the bank of the river, the interface is approximately 55 ft below the land surface (Section 4.1). Beneath the Colorado River, the saltwater interface appears to intersect the river channel bottom.

A cross-sectional view of freshwater heads in the alluvial aquifer near the west bank of the Colorado River can be prepared using measured hydraulic heads and TDS concentrations in wells of varied depth in this area along a line that is roughly orthogonal to the river bank. Such a view, based on data from wells ATP-2S, ATP-2D, PZ2M2, PZ2D, and 0403, is shown in Figure 5–14. The contours of equivalent freshwater head in this cross section are based on computed values of this variable for each of the screened sections in the wells used to prepare the figure. Measured TDS concentrations and freshwater heads are posted at the depths they were measured, and the resulting inferred location of the saltwater interface is drawn.

Although ground water flow directions and magnitudes should not be computed strictly on the basis of freshwater head contours presented in this cross section (Jorgensen et al. 1982; Davies 1987), some general conclusions can be drawn using all the information presented. Extrapolation of the saltwater interface in the direction of the Colorado River indicates that it intercepts the riverbed quite close to the river's west bank. Accordingly, because most water through a porous media system containing both brine and fresh water tends to occur above the interface separating the two (Konikow et al. 1997), it can be logically inferred that nearly all of

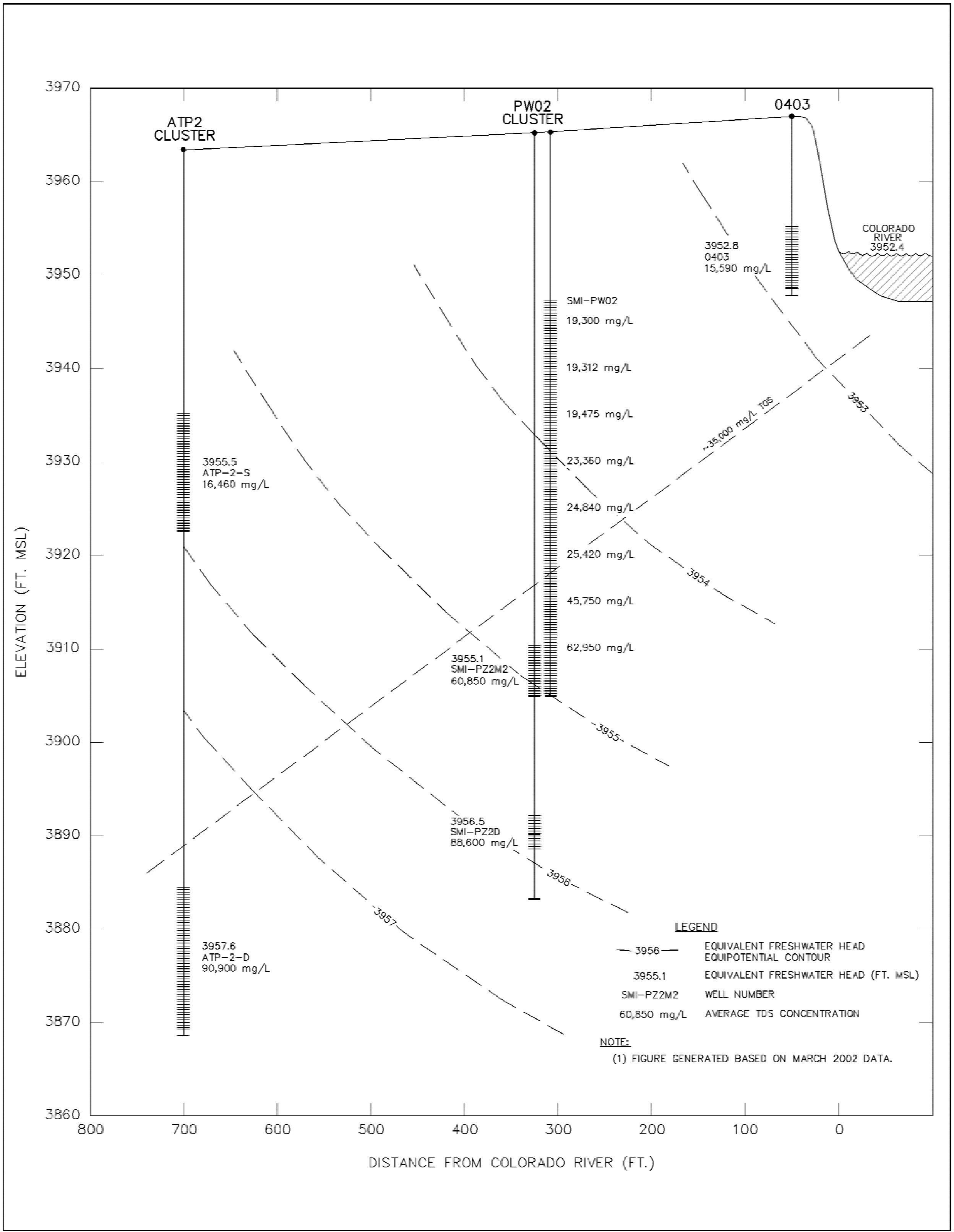

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Figure 5–14. Cross-Sectional View of Equivalent Freshwater Heads and TDS Concentrations in the Alluvial Aquifer near the Colorado River the southeast-moving ground water in this area discharges to the river. This means that both fresh water and briny water are discharging to the river at this location, albeit that ground water flow velocities in the brine are commonly much less than equivalent fresh water velocities, and the volumetric rate of brine discharge is correspondingly lower.

As discussed earlier in this section, the saltwater interface moves laterally and vertically over the course of each year in response to such stresses as seasonal evapotranspiration from the tamarisk community and changes in water stage of the Colorado River. However, because the ground water system at the site appears to be close to a state of dynamic equilibrium, it is expected that local ground water flow will tend to maintain the same general patterns inferred in Figure 5–14.

It is difficult to reliably estimate specific discharges of ground water (i.e., flow vector direction and magnitude) using only freshwater head contours, such as those shown in Figure 5–14 (Davies 1987; Guo and Langevin 2001). However, the freshwater heads in this cross section do suggest that upward flow is occurring locally. The potential for upward flow in the cross-sectional area can be evaluated using two approaches. The first uses data from the two well pairs in the cross section and a simple technique for estimating vertical hydraulic gradients referred to as the Darcy Method (Jorgensen et al. 1982). The second is based on cross-sectional density-dependent flow modeling.

Applying the Darcy Method to water levels and TDS concentrations measured at the ATP2 and PW02 well clusters for the conditions depicted in Figure 5–14 results in respective estimated vertical hydraulic gradients −0.04 and −0.07 both of which indicate upward flow. Similar calculations are applied to several sets of water level and TDS data from the ATP-1 well cluster, which is located west of the Colorado River and north of the area depicted in the cross section. The hydraulic gradients from these latter calculations, presented in Figure 5–15, indicate that upward flow gradients exist at ATP1 during most of the period analyzed. Exceptions occur, however, such as in December 2002. The calculated gradient in one well pair in December 2001 also suggests temporary downward flow at this location.

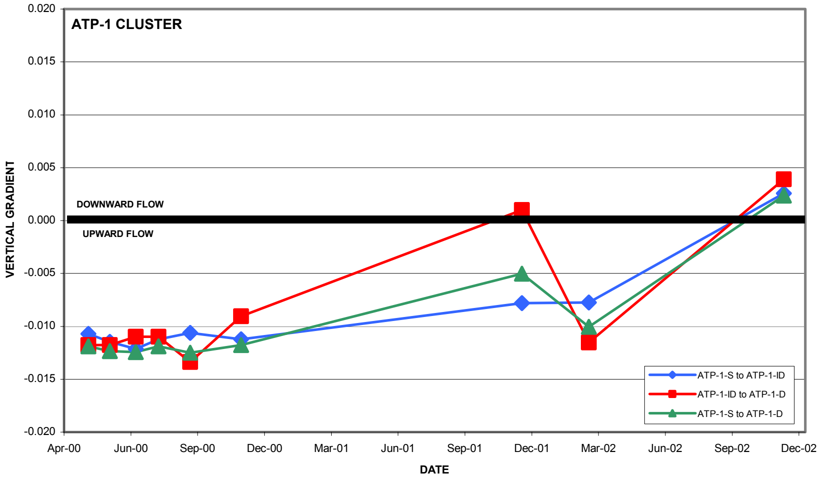

Figure 5–15. Computed Vertical Hydraulic Gradients (accounting for freshwater head and water density) at the ATP-1 Well Cluster in the Alluvial Aquifer

Because spot calculations are not reliable estimators of ground water specific discharge in density-dependent flow systems, numerical models of multidimensional density-dependent flow are often recommended to estimate true flow directions and magnitudes (Jorgensen et al. 1982; Davies 1987). The density-dependent modeling described in Appendix D, Calculation X0062600 helps to demonstrate the flow patterns and relative flow velocities that are expected at the site along the west bank of the Colorado River. The results of the modeling are very similar to the conditions shown in Figure 5–14, in that freshwater head contours indicate a potential for upward flow, and the contour representing a TDS concentration of 35,000 mg/L becomes shallower as it approaches the Colorado River and eventually intersects the riverbed close to the river's west bank. A cross-sectional view of computed pore water velocities with this model (Appendix D, Calculation X0062600) indicates that nearly all of the ground water moving southeast toward the river discharges to the river along a relatively small portion of the total riverbed width, all in an area lying close to the river's west bank. The greatest discharge occurs at the west bank and gradually decreases with distance toward the center of the river. All pore-water velocities west of and just below the river have upward components.

Figure 5–16 illustrates the results of the density-dependent modeling and ancillary information suggesting upward flow in the alluvial aquifer just west of the river and shows the general ground water flow patterns, relative flow magnitudes, and ground water discharges to surface water that can be expected near the west bank of the Colorado River. The vectors in this plot show that ground water flow on the brine side of the saltwater interface is much slower than on the freshwater side. The gradual decrease in velocity vector lengths between the west bank of the river and the river center reflects the tendency for all of the southeast-moving ground water at the site to discharge close to the river's west bank. This also indicates that ground water below the center of the river is expected to move very slowly, signifying that any dissolved contamination below the river is also expected to migrate very slowly. Such flow behavior is dictated by basic principles of ground water hydraulics, and similar flow distributions can be seen in the analogous problem dealing with ground water flow over salt domes (Konikow 1997).

5.2.3 Water Budget

The water budget describes the various components of the hydrologic system at the Moab site and the amounts of water that can reasonably be assigned to them. For the purpose of this discussion, the site environs coincide with the domain of the three-dimensional numerical flow and transport model presented in Section 7.0. This approach facilitates comparison of field water budget values with the model-calculated water budget. A map of the site with the outline of the numerical model projected onto it is presented in Figure 5–17.

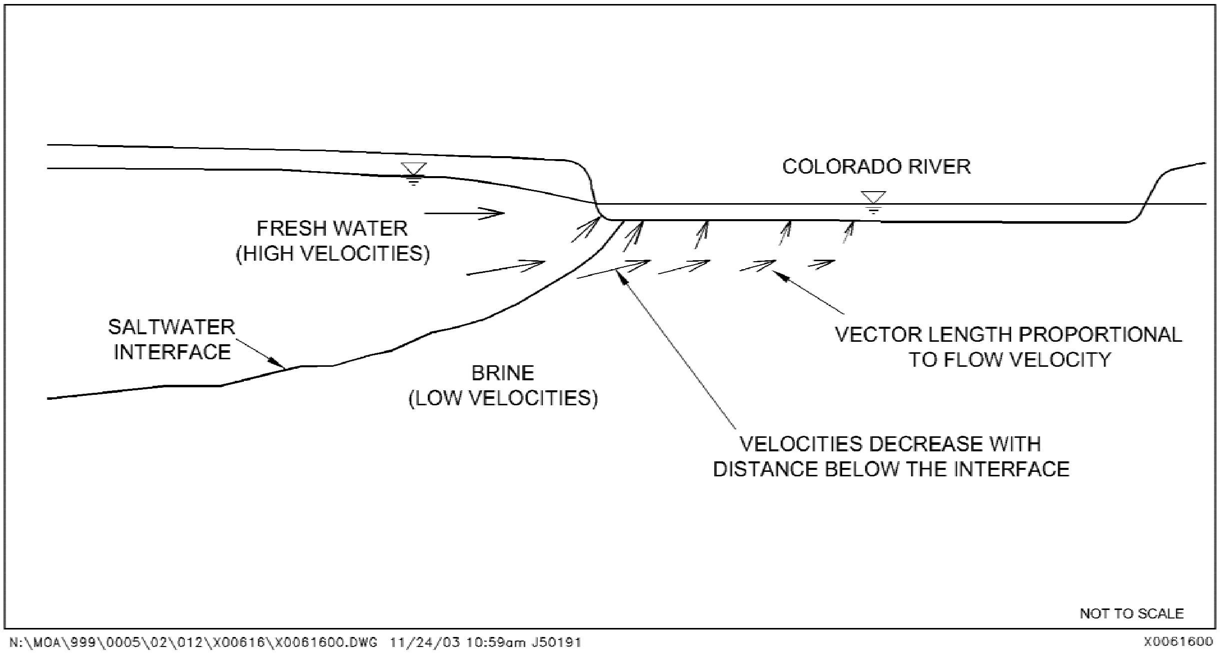

Figure 5–16. General Ground Water Flow Patterns and Relative Velocities Expected near the West Bank of the Colorado River

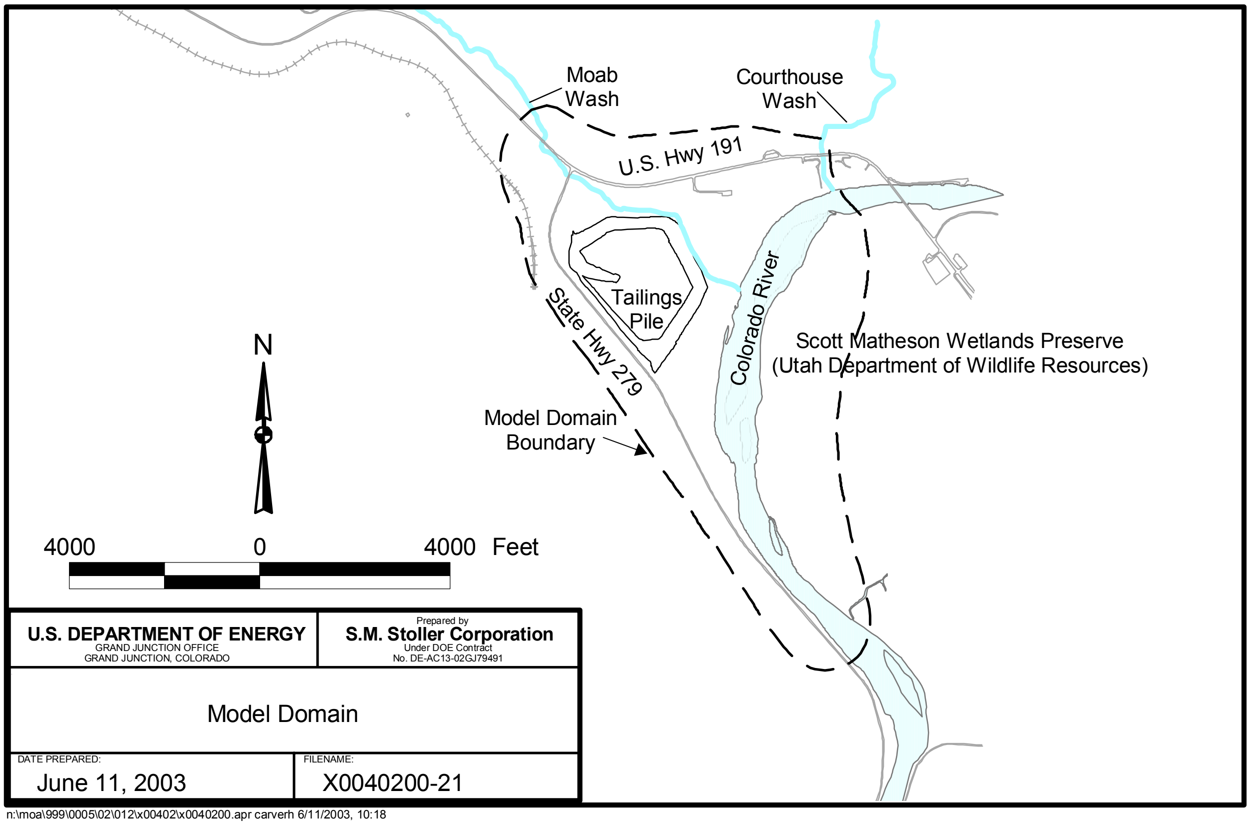

Figure 5–17. Model Domain

5.2.3.1 Inflow

Sources of inflow to the alluvial aquifer include areal recharge from precipitation, direct inflow of ground water from the Glen Canyon aquifer and Entrada Sandstone, and subsurface inflow from Moab Wash. Most of the fresh water in the alluvial aquifer enters the site along geologic contacts between the alluvium and the Glen Canyon aquifer and the Entrada Sandstone in the northwestern and northern parts of the site. Vertical offsets along the Moab Fault truncate the areas where the two bedrock aquifers contact the alluvium; some of the fresh water flowing into alluvium from bedrock likely moves upward within the fault zone. The bedrock in the vicinity of the site is highly fractured and faulted from collapse of the Moab anticline, which was caused by dissolution of underlying Paradox Formation salt beds.

Less significant inflows of fresh water to the alluvial aquifer occur as areal recharge from precipitation and ground water flow beneath Moab Wash. A relatively small quantity of contaminated water enters the alluvial aquifer as a result of seepage of tailings pore fluids from the base of the pile. Inflow of water to the alluvial aquifer from the Colorado River is insignificant because the ground water west of the river almost exclusively discharges to it. The Colorado River occasionally loses water to the alluvial aquifer in the form of temporary bank storage, such as during periods of high runoff. The magnitude of the possible inflow from the Colorado River to the alluvium is unknown.

Areal Recharge

The rate of areal recharge at the site has not been measured. It is estimated to range from 5 to 20 percent of average annual precipitation. The Western Regional Climate Center (www.wrcc.dri.edu/cgi-bin/cliMAIN.pl?utmoab) reports that the average annual precipitation at Moab is 9.09 inches; consequently, the estimated recharge rate ranges from 0.45 to 1.8 inches/year. Excluding the Colorado River and the tailings pile, the surface area of the site (as measured in the numerical model) is approximately 700 acres (Figure 5–17). This area includes exposed bedrock areas where precipitation might be shed as runoff. Application of the potential range in recharge rates to the site area results in a volumetric recharge rate that ranges from 16 to 65 gallons per minute.

Moab Wash

Inflow of ground water in Moab Wash alluvium near the northwestern boundary of the site is estimated using Darcy's Law. Inputs to Darcy's Law are the hydraulic gradient (dh/dl) at the Moab Canyon outlet, the cross-sectional area of the wash (A), obtained by multiplying the approximate wash width of 200 meters by the saturated thickness at well 430 (12.2 m), and the estimated hydraulic conductivity of wash sediments (K = 0.9 to 60 meters/day [m/day]),

The computed inflow obtained by applying the low hydraulic conductivity estimate of 0.9 m/day is

$$Q = K \times A \times (dh/dl) = 0.9 \text{ m/day} \times 2{,}440\text{m}^2 \times (1.2 \times 10^{-3}) = 2.6 \text{ m}^3\text{/day (or 0.5 gpm)}$$

The high inflow estimate based on a hydraulic conductivity of 60 m/day is

$$Q = 60 \text{ m/day} \times 2{,}440 \text{ m}^2 \times (1.2 \times 10^{-3}) = 178 \text{ m}^3\text{/day (or 33 gpm)}$$

Bedrock Aquifers

Inflow to the alluvial aquifer from subcropping bedrock aquifers is also estimated using Darcy's Law. The area of the Glen Canyon aquifer where it contacts basin fill alluvium (Plate 11) is estimated at 1,891,105 ft^2; the estimated subcrop area of the Entrada Sandstone is 1,133,293 ft^2. Water levels measured in the well pair MOA-433 and MOA- 455 indicate an estimated vertical hydraulic gradient between bedrock and alluvium of -3.57×10^{-3}. The assumed hydraulic conductivity of competent rock in both the Glen Canyon Group and Entrada Sandstone is 0.5 ft/day. The resulting estimated inflow to alluvium from both bedrock aquifers is

$$Q = K \times A \times (dh/dz) = 0.5\text{ft/day } (3{,}024{,}400 \text{ ft}^2)(-3.57 \times 10^{-3}) = 5{,}400 \text{ ft}^3\text{/day (or 28 gpm)}$$

Eisinger and Lowe (1999) report that hydraulic conductivities in fractured sections of Navajo Sandstone can be as large as 88 ft/day. Given the likelihood that the Glen Canyon aquifer and Entrada Sandstone are locally quite fractured, bedrock inflow at the Moab site could be considerably larger than estimated above. Assuming that the hydraulic conductivity of the Glen Canyon aquifer and Entrada Sandstone near the Moab Fault could be one order of magnitude larger than the average hydraulic conductivity applied to competent bedrock, combined discharge from the two bedrock aquifer could be as large as 280 gpm. Depending on the degree of fracturing in bedrock, larger inflows are possible.

Tailings Seepage

Inflow from the tailings pile under both current and potential future conditions is estimated in Appendix D, Calculation X0025700, which accounts for precipitation, stored water in the tailings pile, and consolidation of basal slimes. The resulting estimate of current inflow from the pile is 20 gpm. If a no-action alternative is adopted at the site, the tailings drainage rate is expected to eventually decrease to 8 gpm. If a cover designed to restrict the infiltration rate to 1×10^{-8} cm/s is constructed, it is estimated that inflow from the pile will reach a steady value of 0.8 gpm.

5.2.3.2 Outflow

Outflow of water from the alluvial aquifer occurs as discharge to the Colorado River and evapotranspiration by tamarisk.

Evapotranspiration

Estimated rates of evapotranspiration (ET) from tamarisk can be quite large. A DOE study of ET (DOE 2003c) used infrared imaging and vegetation characterization techniques to develop a range of ET rates for tamarisk plants between the tailings pile and the Colorado River. They divided the tamarisk community into three zones of varying stress level (Section 4.10). Using literature values of tamarisk transpiration rates, they estimated that ET rates during the 2002 growing season ranged from 2.5 to 5.0 m/yr for Zone 1, 1.0 to 2.5 m/yr for Zone 2, and 0.0 to 1.0 m/yr for Zone 3. The surface area corresponding to the three zones were 32.6 acres, 20.5 acres, and 33.4 acres, respectively. By multiplying the minimum and maximum estimated ET rates for each zone by the zone areas, estimated ranges of volumetric ET rates were developed, as summarized in Table 5–4.

Table 5–4. Summary of Computed Volumetric ET Rates

Tamarisk Zone	Estimated Volumetric ET Rate (gpm)	
	Low Estimate	High Estimate
1	166	332
2	42	104
3	0	68
Total	208	504

Ground Water Discharge to the Colorado River

Discharge to the Colorado River is approximated using the Dupuit-Forchheimer (DF) discharge formula (Bear 1979). In applying this method, it is assumed that the interface between the brine and overlying relatively fresh water forms a no-flow boundary, which, in the absence of pumping, is fixed. The computed discharges do not take into account any seasonal effects that evapotranspiration from tamarisk plants might have on depth of the interface.

Application of the DF method to the Moab site is similar to previous applications by Glover (1959) and Strack (1989) for estimating freshwater discharge to the ocean in coastal aquifers. In coastal settings, the point of ground water discharge is represented by a fixed sea level; at the Moab site, the Colorado River, which contains fresh water, becomes the point of discharge.

Because the depth from the top of the saturated zone to the saltwater interface depends on proximity to the Colorado River (see Figure 5–11 and Section 5.2.2.4), an estimated average elevation for the saltwater interface must be adopted in the DF calculations. The interface is assumed to be flat and located about 55 ft below the average elevation of the river’s surface.

The DF discharge formula as applied to the Moab site is

$$Q = K\ W\ [(h_0^2 - h_L^2)/2L]$$

where Q = discharge to the river (m^3/day),
K = hydraulic conductivity of the aquifer (m/day),
W = aquifer width orthogonal to flow direction (m),
h_0 = height of water table above the saltwater interface at an upstream location (m),
h_L = height of water table above the saltwater interface at a downstream location (m), and
L = distance between the upstream and downstream locations (m).

To develop a low estimate of aquifer discharge to the river, average water levels in upstream well ATP-2S and downstream well PW-02 (Plate 1) are adopted and an aquifer hydraulic conductivity is 30 m/day is assumed. Given a distance of 122 m between the two wells, the resulting outflow for an area 1,600 m wide is

$$Q = (30\ \text{m/day}(1{,}600\ \text{m})[(193.77\ \text{m}^2 - 185.50\ \text{m}^2)/2\ (122\ \text{m})] = 1{,}627\ \text{m}^3\text{/day (or 300 gpm)}$$

The comparable high flow is computed using a K of 46 m/day

Q= (46 m/day(1,600 m)[(193.77 m^2 – 185.50 m^2)/2 (122 m)] = 2,494 m^3/day (or 460 gpm)

5.2.3.3 Summary of Water Budget

Synthesis of the foregoing estimates of various components of system inflow and outflow results in the annual water budget presented in Table 5–5. This table supports the observation that most of the fresh water in the alluvial aquifer enters the site along geologic contacts between the alluvium and the Glen Canyon Group and Entrada Sandstone bedrock aquifers, which are present beneath the northwestern and northern portions of the site. None of the bedrock aquifer inflow is attributed to flows through the Paradox Formation, since this formation is believed to have very low permeability. Short-term transient effects such as the small contribution to bank storage via losses from the Colorado River during periods of high flow are not reflected in Table 5–5.

Estimated minimum and maximum total inflows to the site are notably less than comparable estimated total outflows. Though this disparity tends to reflect the considerable uncertainty in estimated water budget components, total flows listed in Table 5–5 suggest that the true volume of water moving through the ground water system during an average year could lie somewhere between the maximum total inflow of 400 gpm and the minimum estimated total outflow of 500 gpm. It appears reasonable to assume that an average of 450 gpm passes through the Moab site ground water system.

Table 5–5. Estimated Annual Water Budget for the Moab Site

Flow Component	Inflow (gpm)	Outflow (gpm)
Areal Recharge	16–65	
Moab Wash	0.5–33	
Bedrock Aquifers	28–280	
Tailings Pile	20	
Evapotranspiration		208−504
Colorado River		300–460
Total (rounded)	65−400	500−960

5.3 Background Ground Water Quality

Background water quality is defined as the quality the water would have if uranium-milling activities had not taken place at the site. Background water quality can be inferred by characterizing the water quality in areas that are unaffected by process contamination but are located in the same flow system that influences the Moab site. Regional background ground water quality and local background conditions at the site are evaluated in the following sections.

5.3.1 Regional Water Quality in the Unconsolidated Basin-Fill Aquifer

Ground water in the unconsolidated aquifer is calcium-bicarbonate type or calcium-sulfate-bicarbonate type (Sumsion 1971) and is generally of poorer quality than that of the Glen Canyon and Entrada aquifers. Steiger and Susong (1997) evaluated water sampled from 28 wells completed in the basin-fill and noted that TDS concentrations ranged from 260 to 1,820 mg/L, classifying the water as fresh to slightly saline according to the system described by the USGS

(Robinove et al. 1958) and presented in Table 5–6. About 86 percent of the wells sampled have concentrations less than 1,000 mg/L. Analysis of water samples collected from selected wells completed in the basin-fill indicates that concentrations of arsenic, barium, cadmium, chromium, lead, mercury, selenium, silver, gross alpha, and nitrate are mostly undetectable and, if present, do not exceed the State of Utah water-quality standards or EPA ground water standards listed in Table 2–1. Organic constituents were also analyzed by Steiger and Susong (1997) for some of the samples, and all concentrations were less than detection limits.

Table 5–6. USGS Classification of Water Quality Based on TDS Concentrations

Water Quality Classification	TDS (mg/L)
Fresh	Less than 1,000
Slightly Saline	1,000–3,000
Moderately Saline	3,000–10,000
Very Saline	10,000–35,000
Briny	More than 35,000

Fresh ground water discharging from the Glen Canyon aquifer as seeps and springs mixes with and dilutes TDS concentrations in the basin-fill aquifer along the northeastern margin of Spanish Valley (Sumsion 1971). Mixing of the two water types is irregular but reported as progressively more apparent as the water moves toward the northwest through Spanish Valley and discharges into the Colorado River. Sumsion (1971) noted that although Spanish Valley is underlain at depth by the Paradox salt beds, no saline water or brine was observed in the basin-fill aquifer. This was attributed to impervious shale and several thin anhydrite layers noted on a driller’s log; these strata separate the basin-fill material from the principal evaporite deposits of the underlying Paradox salt beds (Sumsion 1971) near the city of Moab.

More recent sampling of shallow ground water wells completed approximately 10 ft below ground surface in the basin-fill deposits farther to the northwest and closer to the Colorado River than the Sumsion study indicate relatively high concentrations of TDS in the alluvial ground water and suggest the source for the high TDS concentrations is natural and most likely the underlying Paradox salt beds (Cooper and Severn 1994). This sampling was conducted on the south side of an inside meander of the Colorado River just west of the U.S. Highway 191 bridge in the Matheson Wetlands Preserve. Two major water types reported to occur in the wetlands area are a calcium sulfate type dominated by gypsum salt and a sodium chloride (salt) water. The highest TDS concentrations, ranging from 21,800 to 44,400 mg/L, were measured in water samples collected from three wells (M9, M10, and M11) located near the Colorado River at the northernmost end of the preserve (Plate 1). These samples also had the highest sodium (7,590 mg/L) and chloride (20,700 mg/L) concentrations and the lowest sulfate-to-chloride concentration ratios (<1). The relatively high TDS concentrations classify the ground water at these locations as very saline to briny (Table 5–6).

Water analysis of major ions and trace metals from the Cooper and Severn (1994) study indicate that the shallow alluvial ground water near the Colorado River in the wetlands area is of poor quality. Naturally occurring concentrations of arsenic, cadmium, and lead exceed EPA standards in 40 CFR 192 in samples from all three wells (M9, M10, and M11). Naturally occurring concentrations of molybdenum exceed the EPA standard in one sample from location M11. Maximum detected concentrations of arsenic (0.12 mg/L), cadmium (0.019 mg/L), molybdenum

(0.101 mg/L), and lead (0.18 mg/L) are all slightly higher than the 40 CFR 192 standards of 0.05 mg/L, 0.01 mg/L, 0.10 mg/L, and 0.05 mg/L, respectively. Sulfate concentrations all greatly exceed the secondary drinking water standard of 250 mg/L. Concentrations are also higher than the secondary standards for chloride (250 mg/L) and manganese (0.05 mg/L). The secondary standard for iron (0.3 mg/L) is exceeded at location M10. Although secondary standards are not enforceable or health-based levels, the fact that levels are exceeded provides additional evidence of the naturally poor quality of alluvial ground water.

5.3.2 Regional Water Quality in the Consolidated Bedrock Aquifers

The Glen Canyon Group contains the principal bedrock aquifer in the region. Other consolidated rocks, such as the Entrada Sandstone, are capable of transmitting and yielding small quantities of water but are not important as water resources (Sumsion 1971), and consequently, few wells are completed in the Entrada aquifer in the Moab and Spanish Valley region. However, the aquifer commonly does yield fresh water in quantities of 5 gpm or less to seeps and small springs at the contact between the Slick Rock Member and the less permeable underlying Dewey Bridge Member (Blanchard 1990). One of the primary spring discharge areas is in the canyon walls of Courthouse Wash. Blanchard (1990) analyzed water samples collected from several springs and from a flowing well completed in the Entrada aquifer and reported that concentrations of arsenic, barium, iron, and selenium were less than EPA ground water standards listed in Table 2–1. Water quality from the Entrada aquifer is higher in and near areas where the Entrada Sandstone crops out in Moab and Spanish Valleys. The water is hard to very hard, and TDS concentrations typically range from about 100 to 500 mg/L (Blanchard 1990), classifying the water as fresh (Table 5–6). Farther north of Moab and Spanish Valleys where the sandstone is more deeply buried, TDS concentrations have been observed to increase to 4,330 to 104,000 mg/L (Blanchard 1990), classifying the water as moderately saline to briny.

The Glen Canyon aquifer yields water to wells where the sandstones are faulted and highly fractured. Water quality is excellent and is characterized by a calcium-magnesium-bicarbonate water type that is typically hard (Blanchard 1990). A broad evaluation of the regional ground water from samples collected at 28 springs and wells completed in the Glen Canyon aquifer in the Spanish Valley area (Steiger and Susong 1997) indicate concentrations of arsenic, barium, cadmium, chromium, lead, mercury, selenium, silver, gross alpha activity, and nitrate do not exceed EPA ground water standards listed in Table 2–1. Organic constituents were analyzed in some of the samples, and concentrations were less than State of Utah water-quality standards. TDS was generally less than 500 mg/L, classifying the water as fresh (Table 5–6).

The city of Moab derives most of its drinking water from a well field completed in the Glen Canyon aquifer near the northeast canyon wall of Spanish Valley (Blanchard 1990). Two water-supply wells located at the NPS headquarters near the entrance to Arches National Park are completed in the Navajo Formation. Blanchard (1990) notes that the water quality from the Navajo aquifer at this location is different from that in the Navajo aquifer at other locations. Relatively higher TDS concentrations in the Arches wells, ranging from 584 mg/L to 1,000 mg/L, are attributed to the proximity of the Moab Fault (Blanchard 1990).

5.3.3 Regional Water Quality in the Paradox Formation

The greatest impact to the water quality in the basin-fill aquifer prior to milling operations is due to natural dissolution of salt beds in the Paradox Formation that are present in subcrops throughout most of the site (Figure 5–18) and Spanish Valley. Mayhew and Heylman (1965) reported that supersaturated brines, containing substantial quantities of trace elements, are present in the Moab region where the Paradox salts are well developed. Results for selected Paradox Formation brine samples that were collected from three oil and gas exploration wells located close to the Moab site are summarized in Table 5–7. Oil and gas well Delhi-Taylor No. 2 is located approximately 4 miles northwest of the site, Southern Natural Gas No. 1 Long Canyon is located approximately 7 miles southwest of the site, and King Oil No. 2 Big Flat is located approximately 11 miles southwest of the site. Results for two samples collected from different intervals from the Paradox brine are reported for the Southern Natural Gas No. 1 well. Analyses were performed by chemists from USGS and commercial laboratories.

Table 5–7. Results of Analysis of Brine Samples Collected from the Paradox Formation in the Moab Region

Analyte	**Concentration (mg/L)**			
	Delhi No. 2	**King Oil**	**So Nat Gas**	**So Nat Gas**
Aluminum	66			
Ammonia	849	1,330		
Bicarbonate	1,010		1,600	1,400
Borate		2,922		
Boron	660		600	
Bromine	3,080	1,150	3,000	6,100
Calcium	52,700	40,742	34,000	65,800
Chloride	241,000	259,106	45,000	29,800
Copper	6			
Fluoride	25			
Iodine	42		300	
Iron	750			
Lithium		173		500
Magnesium		47,789	21,000	45,500
Nitrate				6
Phosphate			2,000	
Potassium		41,957		23,400
Rubidium				700
Sodium		25,966	13,000	9,800
Sulfate		754	1,800	80
TDS		421,889	388,000	439,000
pH			4.8 s.u.	6.0 s.u.

Source: Mayhew and Heylman, 1965.

Elevated concentrations of numerous elements, including aluminum, boron, calcium, chloride, copper, iron, lithium, magnesium, rubidium, and potassium are present in the supersaturated brines collected from the Paradox Formation as evidenced by the data presented in Table 5–7. Ammonia and nitrate, both of which are constituents of processes fluids during the milling operation at the Moab site, are present naturally in the brine. Ammonia was detected at relatively high concentrations ranging from 849 to 1,330 mg/L. Nitrate was detected in one sample at 6 mg/L. TDS concentrations average more than 400,000 mg/L. Radionuclides were not analyzed in the brine samples; however, a gamma log obtained for the Southern Natural Gas No. 1 well indicates that a gamma anomaly is associated with a black, fetid shale unit in the Paradox Formation. The anomalous gamma signal suggests that naturally occurring concentrations of uranium and radium associated with the carbonaceous shale may also be present.

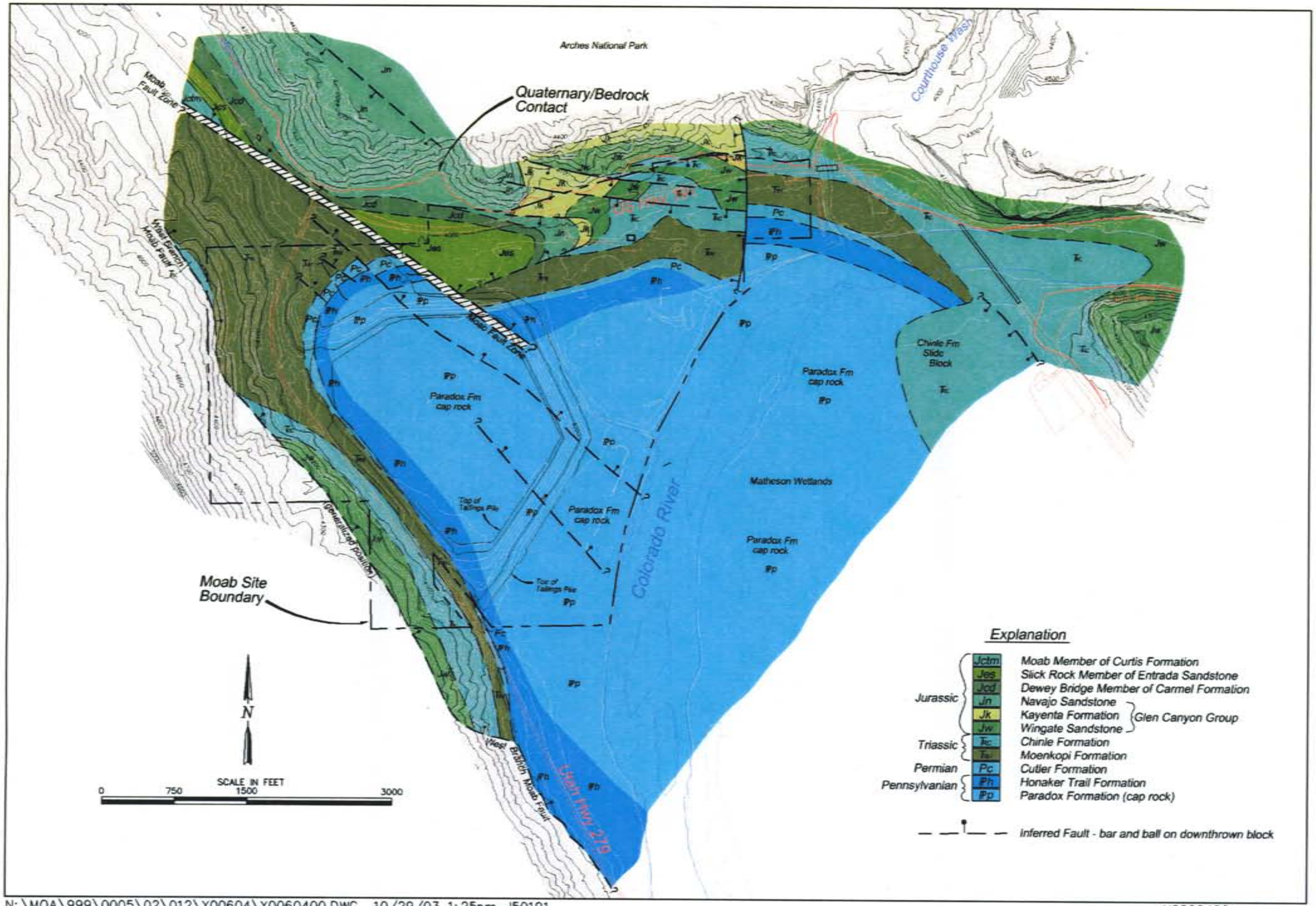

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Figure 5–18. Geologic Subcrop Map

Formation pressures in some of the oil and gas exploration wells were sufficiently high that concentrated brine flowed to the surface (Mayhew and Heylman 1965). High formation pressures could provide another mechanism, in addition to dissolution, for concentrated brine to enter the basin-fill aquifer in areas where the Paradox Formation is fractured.

5.3.4 Site Background Water Quality

The most direct method to characterize background water quality is to analyze water samples collected from wells located on the site prior to milling operations. Historical records indicate that two water supply wells were present at the site prior to the start of milling operations in 1956 (see Table 3–11). Both wells were located near the northwest area of the former pile (Plate 1). Records indicate the first well, designated as well C, was installed to a depth of 67 ft by the U.S. Department of Interior Grazing Service in 1940 and provided approximately 20 gpm from the unconfined aquifer, probably for livestock. The second well, designated as well B, was installed to a depth of 114 ft by the AEC in 1954 just prior to mill construction. The zone of completion for well B is unknown. This well produced approximately 11 gpm through a perforated casing and was presumably used to supply process water for the mill. In both cases, the quality of the water is unknown and the wells have subsequently been abandoned. No other water wells are known to have existed at the site prior to milling.

Upgradient fresh water enters the site from Moab Wash and along geologic contacts between the alluvium and the Glen Canyon Group at the northern boundary of the site, especially where the Moab Fault has highly fractured these bedrock units. A conceptual model of the subsurface hydrogeology along a representative streamline showing the saltwater interface between the deeper brine system and where the overlying fresh water enters the system is illustrated in Figure 5–11. Fresh water entering the site at the northern boundary flows toward the river in a southerly direction over the top of a deeper, natural brine zone. The deeper brine water results mostly from dissolution of the underlying salt beds of the Paradox Formation present beneath most of the site. The extensive area where the basin-fill aquifer is in contact in the subsurface with the Paradox Formation is shown by the geologic subcrop map in Figure 5–18. Chinle and Moenkopi Formations shown in Figure 5–18 near the northern boundary of the site may also contribute some saline water to the basin-fill alluvium. Mixing of the two background water types—fresh upgradient water with the deeper saline water—influences the background water quality at the site. The result is a background water quality that is highly stratified both vertically and horizontally across the site.

Vertical and horizontal variability in the water quality is reflected by the distribution of TDS concentrations posted in Figure 5–19. These data represent results of the most recent ground water sampling of permanent and temporary wells (Section 4) that are summarized in Appendix C. In a few cases, earlier data collected in 2000 by SMI (Section 3.0) are used to provide coverage in areas of the site where the monitor well was decommissioned before DOE could sample the location. These areas are mostly related to the temporary wells installed by SRK (2000) in the basin-fill aquifer beneath the tailings pile and the temporary wells installed by ORNL (1998a) along the bank of the Colorado River.

Environmental Visualization System (EVS) three-dimensional modeling software (http://www.ctech.com) was used to krige and contour the data in Figure 5–19. The EVS three-dimensional model allows visualization of the contoured surface at any discrete elevation in the

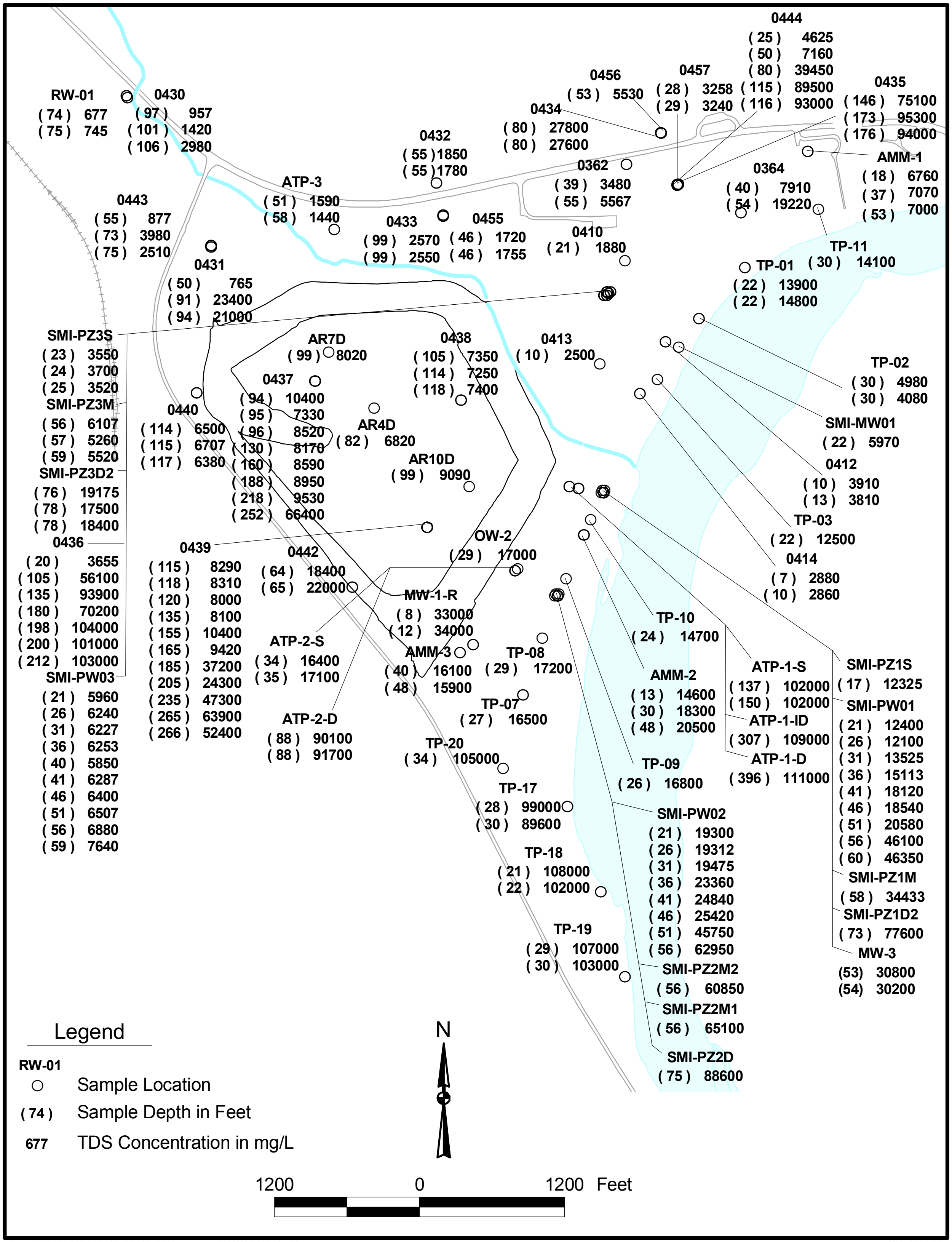

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Figure 5–19. TDS Concentrations in the Alluvial Aquifer as a Function of Depth horizontal or vertical plane. The areal distribution of the TDS concentrations, as represented by the contours on the upper surface of the ground water (first water), is presented in Figure 5–20. A cross-sectional view of the contoured TDS concentrations beginning at Moab Wash and extending southeast to the Colorado River is shown in Figure 5–21. It is apparent from the figures that the relatively low TDS concentrations of less than 10,000 mg/L reflect the influence of the fresh water entering the site from Moab Wash and from the fractured water-bearing bedrock formations at the northern boundary. TDS concentrations greater than 35,000 mg/L in ground water reflect the influence of saline water derived from the deeper Paradox Formation. The interface between the upper fresh water and the deeper saline water is sharp and extends across the site in a wedge shape, in which the deepest part of the interface is near the northwest boundary of the site and the shallowest depth is near the river.

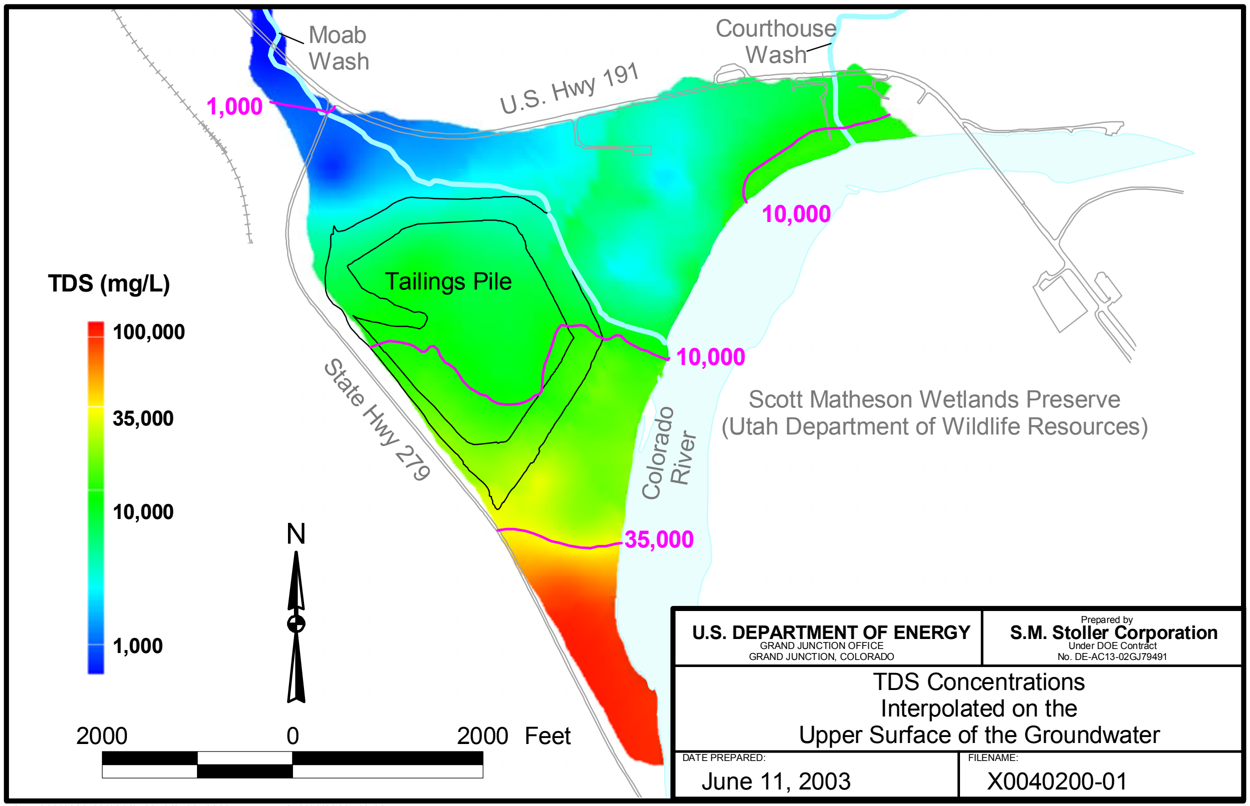

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Figure 5–20. Areal Distribution of the TDS Concentrations Contoured on the Upper Surface of the Ground Water

5.3.4.1 Water Quality Influence from the Glen Canyon Aquifer

Fresh water from the Glen Canyon Group enters the basin-fill aquifer at the site by subsurface inflow in two ways. The first is indirectly through recharge to Moab Wash (Qmd) and the second is through direct recharge in areas north of the site where the Glen Canyon Group underlies the basin-fill aquifer and where it is highly fractured along the Moab Fault zone. ARCHES1978 is a water supply well installed by the NPS in 1978 approximately 2 miles northwest of the site near the entrance to Arches National Park and is completed in the Navajo aquifer. Water from ARCHES1978 is considered representative of the water from the Navajo aquifer that provides recharge to the alluvial aquifer within Moab Wash.

Monitor well MOA-432, also completed in the Navajo aquifer, is located near the northern boundary of the site where the rocks are similarly faulted and fractured. Water from MOA-432 is considered representative of the water quality from the Navajo aquifer where the sandstones provide direct recharge to the basin fill aquifer underneath the site. A significant amount of the total freshwater recharge to the basin-fill aquifer in the northern area of the site is associated with upward and lateral flow from the Glen Canyon aquifer where the Moab Fault zone and the water-bearing bedrock formations are in contact with the alluvium (see Section 5.2).

Water quality results obtained on samples collected from ARCHES1978 and MOA-432 are summarized in Table 5–8 and presented in the Piper diagram in Figure 5–22. This background water type has a sodium cation composition with mixed anionic species. TDS concentrations range from 794 to 1,850 mg/L, which classifies the water as fresh to slightly saline (Table 5–6). Maximum concentrations of arsenic, cadmium, chromium, lead, mercury, molybdenum, nitrate, selenium, silver, radium-226, radium-228, and uranium (Table 5–8) do not exceed EPA standards in 40 CFR 192. Background alkalinity, as calcium carbonate, ranges from 139 to 203 mg/L. Sulfate concentrations range from 225 to 342 mg/L. Calcium concentrations range from 84 to 95 mg/L. Magnesium ranges from 35 to 51 mg/L. On average, the Navajo Aquifer water pH is near neutral (7.45), and the redox condition is slightly oxidizing (ORP 68 millivolts [mV]).

5.3.4.2 Water Quality Influence from Moab Wash

Fresh alluvial ground water from Moab Wash enters the site at the northwest boundary and is represented by water samples collected at monitor well RW-01, which is located approximately 1,650 ft upgradient from the site. Analytical results on water samples collected at RW-01 are summarized in Table 5–8. Water quality results presented in the Piper diagram (Figure 5–22) indicate this water has a sodium cation composition with a mixed anionic species. RW-01 and ARCHES1978 ground water are similar in composition, as indicated by their proximity on the Piper diagram. This evidence suggests the Navajo aquifer provides recharge to the alluvial system in Moab Wash.

TDS concentrations in well RW-01 average 708 mg/L, which classifies the water as fresh (Table 5–6). Background alluvial ground water from RW-01 is further characterized by an average alkalinity as calcium carbonate of 163 mg/L. Sulfate concentrations average 193 mg/L, calcium concentrations average 39 mg/L, and magnesium concentrations average 23 mg/L. On average, the alluvial water pH is near neutral (7.8), and the redox condition is slightly oxidizing (oxidation-reduction potential 275 mV). Maximum concentrations of arsenic, cadmium, chromium, lead, mercury, molybdenum, nitrate, selenium, silver, radium-226, radium-228, and uranium do not exceed EPA standards (Table 2–1).

5.3.4.3 Water Quality Influence from the Chinle and Moenkopi Bedrock Formations

Upward flow gradients from the Chinle and Moenkopi Formations that are present as subcrops in the northern portion of the site (see Section 5.2) contribute a small amount of salinity to the basin-fill aquifer and thus have influenced the water quality at the site prior to milling operations. Permeable layers in these bedrock formations are known to yield very saline water (Figure 5–9). Clay, silt, and gypsum in the Chinle Formation and the common presence of gypsum and salt crystals in the Moenkopi Formation are sources of high concentrations of TDS, chloride, and sulfate (Rush et al 1982).

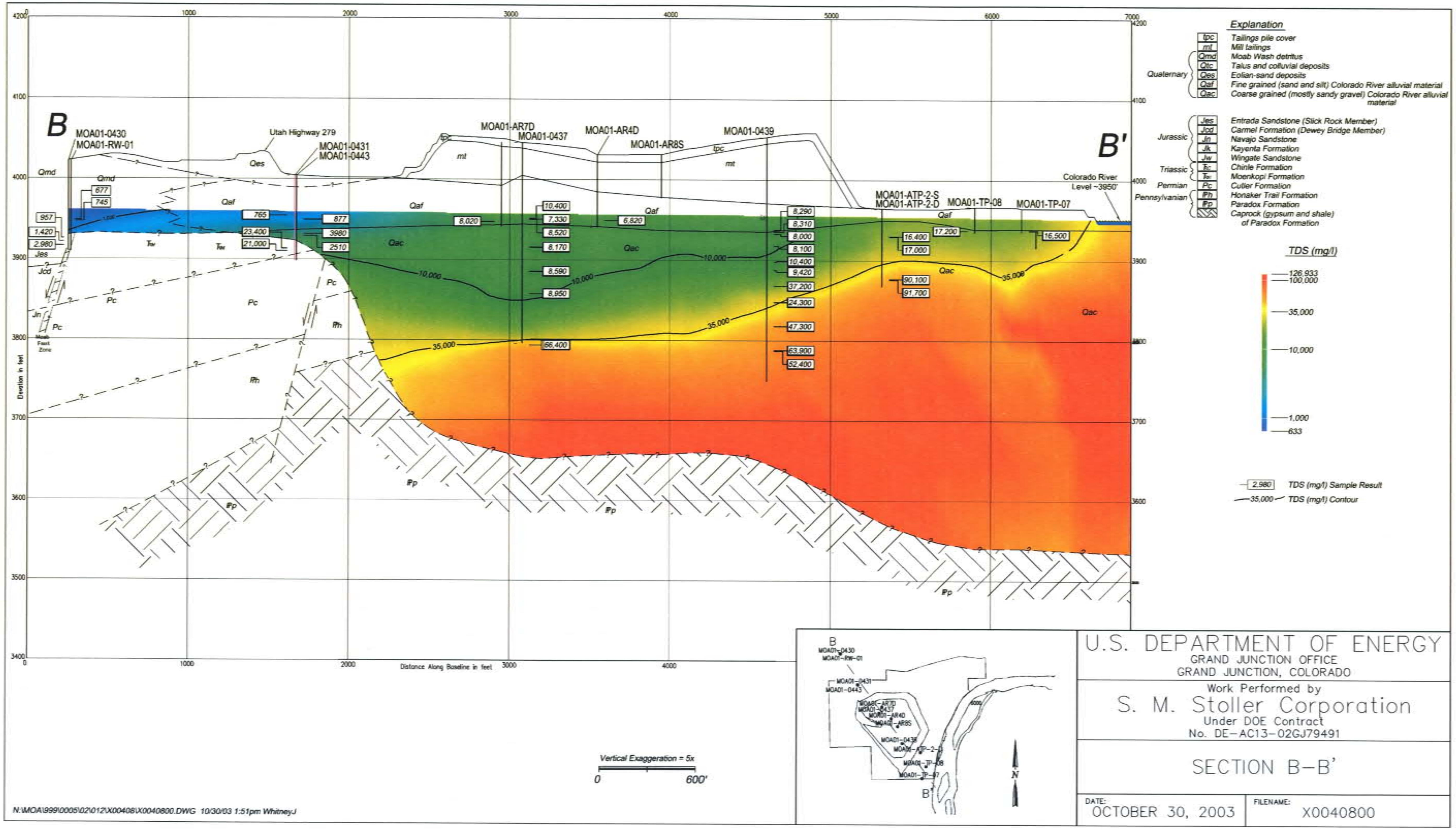


Figure 5–21. Cross-Sectional View of TDS Concentrations in the Moab Wash (Qmd) and Basin-Fill Aquifer (Qaf and Qac) Beginning at Background Location RW-01 and Extending Southeast to the Colorado River

Table 5–8. Background Ground Water Quality for Wells Completed in Bedrock Formations and the Unconfined Alluvial Aquifer Upgradient of the Moab Site

Analyte	Unit	Bedrock Wells									Alluvial Wells								
		Navajo Formation						Chinle Formation			Moab Wash			Basin-Fill					
		ARCHES1978			MOA-432			MOA-434			RW-01			AMM-1			MOA-456		
		Range	Mean	FOD	Range	Mean	FOD	Range	Mean	FOD	Range	Mean	FOD	Range	Mean	FOD	Range	Mean	FOD
Major																			
Alkalinity, total as $CaCO_3$	mg/L	185−203	196.7	3/3	139−151	145	2/2	177−201	189	2/2	147−189	163	3/3	137−157	146.8	6/6	171−171	171	1/1
Ammonia, total as N	mg/L	<0.05−<0.05	0.025	0/2	<0.003−0.0496	0.02555	1/2	0.0854−0.205	0.1452	2/2	<0.003−<0.003	0.0015	0/3	<.003−.038	0.00763	1/6	<0.003−<1	0.2508	0/2
Chloride	mg/L	64−260	155	3/3	729−738	733.5	2/2	15,700−15,800	15,750	2/2	135−174	152.3	3/3	3,250−4,050	3,453	8/8	2,340−2,390	2,365	2/2
Magnesium	mg/L	35−38.1	36.03	3/3	51.4−51.4	51.4	1/1	245−245	245	1/1	<30.7−31.4	23.38	1/2	124−188	145.7	6/6			
Nitrate as NO_3	mg/L	3.43−6.46	4.945	2/2	1.01−1.07	1.04	2/2	<0.02−<0.02	0.01	0/2	13.8−15.9	14.63	3/3	1.22−6.3	2.346	5/5	5.76−5.76	5.76	1/1
Potassium	mg/L	8−9.6	8.533	3/3	18.2−18.2	18.2	1/1	199−199	199	1/1	9.29−10.1	7.17	1/2	49.7−56	48.15	5/6			
Sodium	mg/L	120−191	143.7	3/3	444−476	460	2/2	8,910−9,490	9,200	2/2	<128−140	113.3	2/3	1,920−2,080	1,993	8/8	1,570−1,570	1,570	1/1
Sulfate	mg/L	225−273.7	243.7	3/3	326−342	334	2/2	1,030−1,530	1,280	2/2	180−214	193	3/3	933−1,080	991.8	8/8	1,120−1,140	1,130	2/2
Metals																			
Aluminum	mg/L	<0.03−<0.03	0.015	0/2	<0.0076−<0.0076	0.0038	0/1	<0.038−<0.038	0.019	0/1	<0.0076−<0.0076	0.0038	0/1	<0.0076−<0.051	0.01143	0/3			
Antimony	mg/L	<0.003−<0.003	0.0015	0/2	<0.0039−<0.0039	0.00195	0/1	<0.0001−<0.0001	0.00005	0/1	<0.0001−<0.00019	0.00007	0/2	<0.00019−<0.011	0.00128	0/5			
Arsenic	mg/L	<0.005−<0.005	0.0025	0/3	<0.0001−<0.0001	0.00005	0/2	<0.0001−0.0004	0.00023	1/2	0.0012−0.0015	0.00137	3/3	0.00036−0.0012	0.00054	4/7	0.00018−0.00018	0.00018	1/1
Barium	mg/L	0.025−0.033	0.028	3/3	0.0211−0.0211	0.0211	1/1	0.0675−0.0675	0.0675	1/1	0.0285−0.033	0.03075	2/2	0.0222−0.0323	0.02523	4/4			
Cadmium	mg/L	<0.001−<0.001	0.0005	0/3	<0.0001−<0.00026	0.00009	0/2	<0.0001−<0.00037	0.00012	0/2	<0.0001−<0.00025	0.00008	0/3	<0.0001−<0.0017	0.00019	0/7	<0.0001−<0.0001	0.00005	0/1
Calcium	mg/L	84−94.6	88.2	3/3	94.5−94.5	94.5	1/1	477−477	477	1/1	<47.4−54.6	39.15	1/2	209−294	230.3	6/6			
Chromium	mg/L	<0.005−<0.005	0.0025	0/3	<0.0005−<0.0005	0.00025	0/1	<0.0025−<0.0025	0.00125	0/1	<0.00093−0.0011	0.00078	1/2	<0.0005−<0.011	0.00135	0/5			
Cobalt	mg/L				<0.0013−<0.0013	0.00065	0/1	<0.0065−<0.0065	0.00325	0/1	<0.0013−<0.0013	0.00065	0/1	<0.0013−0.002	0.00155	2/3			
Copper	mg/L	<0.012−<0.012	0.006	0/3	<0.0004−<0.0004	0.0002	0/1	<0.002−<0.002	0.001	0/1	<0.00043−<0.0006	0.00026	0/2	<0.0004−0.005	0.00086	1/5			
Iron	mg/L	<0.02−0.092	0.04357	2/3	0.0251−0.0555	0.0403	2/2	0.52−1.58	1.05	2/2	<0.0019−<0.0131	0.00317	0/3	<0.0008−<0.05	0.00526	0/7	0.0194−0.0194	0.0194	1/1
Lead	mg/L	<0.003−<0.003	0.0015	0/3	<0.00078−	0.00039	0/1	<0.00076−	0.00038	0/1	<0.0001−<0.00077	0.00022	0/2	<0.0001−<0.0055	0.00072	0/5			
Lithium	mg/L				0.0179−0.0186	0.01825	2/2	0.222−0.235	0.2285	2/2	0.0278−0.0418	0.0348	2/2	0.0714−1	0.1814	3/4	0.0457−0.0457	0.0457	1/1
Manganese	mg/L	<0.005−<0.005	0.0025	0/3	0.0017−0.0061	0.0039	2/2	1.44−2.89	2.165	2/2	<0.0001−<0.0005	0.00014	0/3	<0.0001−<0.01	0.00142	0/8	0.0157−0.0157	0.0157	1/1
Mercury	mg/L	<0.0002−<0.0002	0.0001	0/2							<0.0002−<0.0002	0.0001	0/1	<0.0001−<0.0002	0.00009	0/4			
Molybdenum	mg/L	0.002−0.002	0.002	2/2	<0.0018−<0.009	0.0027	0/2	<0.009−<0.009	0.0045	0/2	<0.0018−<0.009	0.0021	0/3	<0.0018−0.01	0.00472	4/8	<0.009−<0.009	0.0045	0/1
Nickel	mg/L	<0.01−<0.01	0.005	0/3							<0.0008−<0.0008	0.0004	0/1	<0.0006−0.015	0.00403	1/4			
Selenium	mg/L	0.004−0.0093	0.00643	3/3	0.0018−0.0018	0.0018	2/2	<0.0001−0.00029	0.00017	1/2	0.0091−0.0097	0.00947	3/3	0.0107−0.015	0.01309	7/7	0.0266−0.0266	0.0266	1/1
Silver	mg/L	<0.002−<0.002	0.001	0/3	<0.00032−	0.00016	0/1	<0.00034−	0.00017	0/1	<0.00025−	0.00014	0/2	<0.0001−<0.0055	0.00068	0/5			
Strontium	mg/L				2.51−2.59	2.55	2/2	12.2−12.7	12.45	2/2	<2.25−2.43	1.965	2/3	5.3−8.45	5.974	7/7	3.44−3.44	3.44	1/1
Thallium	mg/L	<0.001−<0.001	0.0005	0/2	<0.00069−	0.00035	0/1	<0.0022−<0.0022	0.0011	0/1	<0.0001−<0.00069	0.0002	0/2	<0.0001−<0.011	0.0012	0/5			
Uranium	mg/L				0.0018−0.0019	0.00185	2/2	0.0149−0.0174	0.01615	2/2	0.0111−0.012	0.0115	3/3	0.0042−0.0259	0.00806	8/8	0.0166−0.0202	0.0184	2/2
Vanadium	mg/L	<0.04−<0.04	0.02	0/2	0.00046−0.002	0.00073	1/2	0.0017−0.002	0.00135	1/2	<0.0141−0.0164	0.01308	2/3	0.00061−0.011	0.00194	5/8	<0.002−<0.002	0.001	0/1
Zinc	mg/L	0.087−0.16	0.1235	2/2	<0.0008−<0.0008	0.0004	0/1	<0.004−<0.004	0.002	0/1	<0.0006−<0.0008	0.00035	0/2	<0.0006−0.011	0.00426	3/5			
Other																			
Boron	mg/L				0.117−0.122	0.1195	2/2	0.537−0.54	0.5385	2/2	0.106−0.116	0.111	2/2	0.397−0.477	0.437	5/5	0.314−0.314	0.314	1/1
Dissolved Oxygen (unfiltered)	mg/L				3.93−4.23	4.08	2/2	0.4−0.58	0.49	2/2	5.05−6.94	5.995	2/2	0.9−3.99	2.533	4/4	2.41−2.41	2.41	1/1
Fluoride	mg/L	0.421−0.424	0.4225	2/2	<0.108−0.489	0.2715	1/2	<0.502−3.08	1.666	1/2	0.313−0.319	0.316	2/2	<0.193−1.46	0.7494	2/4	0.546−0.546	0.546	1/1
Oxidation Reduction Potential	mV				7.7−129	68.35	2/2	−526−118	−322	2/2	91−619	275.3	3/3	66.9−242	176.8	7/7	107−107	107	1/1
Total Dissolved Solids	mg/L	794−966	858.7	3/3	1,780−1,850	1,815	2/2	27,600−27,800	27,700	2/2	677−745	708	3/3	6,760−7,820	7,113	7/7	5,530−5,530	5,530	2/2
pH (unfiltered)	s.u.				7.4−7.5	7.45	2/2	6.98−7.21	7.095	2/2	7.67−8.11	7.823	3/3	7.19−7.91	7.47	10/10	7.73−7.73	7.73	1/1
Physical Measurements																			
Specific Conductance (unfiltered)	µmhos/cm				2,814−2,924	2,869	2/2	37,835−40,040	38,940	2/2	1,078−1,152	1,120	3/3	8,198−14,040	11,260	9/9	8,381−8,381	8,381	1/1
Specific Gravity					1−1	1	1/1	1.02−1.02	1.02	1/1	1−1	1	1/1	1.01−1.01	1.01	2/2	1.01−1.01	1.01	1/1
Temperature (unfiltered)	°C				19.36−20.59	19.98	2/2	17.46−21.05	19.26	2/2	17.38−19.14	18.46	3/3	14.3−19.4	17.8	8/8	17.9−17.9	17.9	1/1
Turbidity (unfiltered)	NTU				1.26−5.32	3.29	2/2	1.66−2.66	2.16	2/2	5.64−9.67	7.753	3/3	0.2−8.1	2.427	6/6	144−144	144	1/1
Radionuclides																			
Gross Alpha	pCi/L	6−24	13.07	3/3	<12.77−<18.45	7.805	0/2	<205.35−	136.2	0/2	<6.73−7.64	4.862	1/3	<48.96−<73.92	29.24	0/6	<37.38−<37.38	18.69	0/1
Gross Beta	pCi/L	<10−17	11.97	2/3	<16.17−19.84	13.96	1/2	221.35−321.03	190.9	1/2	7.39−17.37	12.16	3/3	<39.89−64.4	29.71	1/6	45.77−45.77	45.77	1/1
Lead-210	pCi/L				<1.14−<1.14	0.57	0/1	<1.23−<1.23	0.615	0/1	<1.12−<1.17	0.5725	0/2	<1.12−<1.46	0.6463	0/4			
Polonium-210	pCi/L				0.0527−0.0527	0.0527	1/1	0.0554−0.0554	0.0554	1/1	<0.0301−<0.0455	0.0189	0/2	<0.035−0.107	0.04551	1/4			
Radium-226	pCi/L	<0.5−<0.5	0.25	0/3	0.14−0.18	0.16	2/2	2.14−3.11	2.625	2/2	<0.1−<0.11	0.05333	0/3	.07−.16	0.07667	2/6	<0.12−<0.12	0.06	0/1
Radium-228	pCi/L	<1−1	0.6667	1/3	<0.71−<0.73	0.36	0/2	0.77−2.07	1.42	2/2	<0.64−<0.7	0.34	0/3	<.5−1	0.4575	1/6	<0.7−<0.7	0.35	0/1
Radon-222 (unfiltered)	pCi/L				<128.01−	64.01	0/1	<498.29−	249.1	0/1	431.46−431.46	431.5	1/1	<392.82−	196.4	0/1			
Thorium-230	pCi/L				<1.5−<1.5	0.75	0/1	<1.5−<1.5	0.75	0/1	<1.5−<1.6	0.775	0/2	<1.5−<1.6	0.7875	0/4			
Uranium-234	pCi/L				1.4−1.4	1.4	1/1	8.7−8.7	8.7	1/1	4.2−10.7	4.775	1/2	5.6−6.1	5.85	2/2			
Uranium-238	pCi/L				0.76−0.76	0.76	1/1	8.1−8.1	8.1	1/1	4.7−4.7	4.7	2/2	1.8−3.2	2.5	2/2			

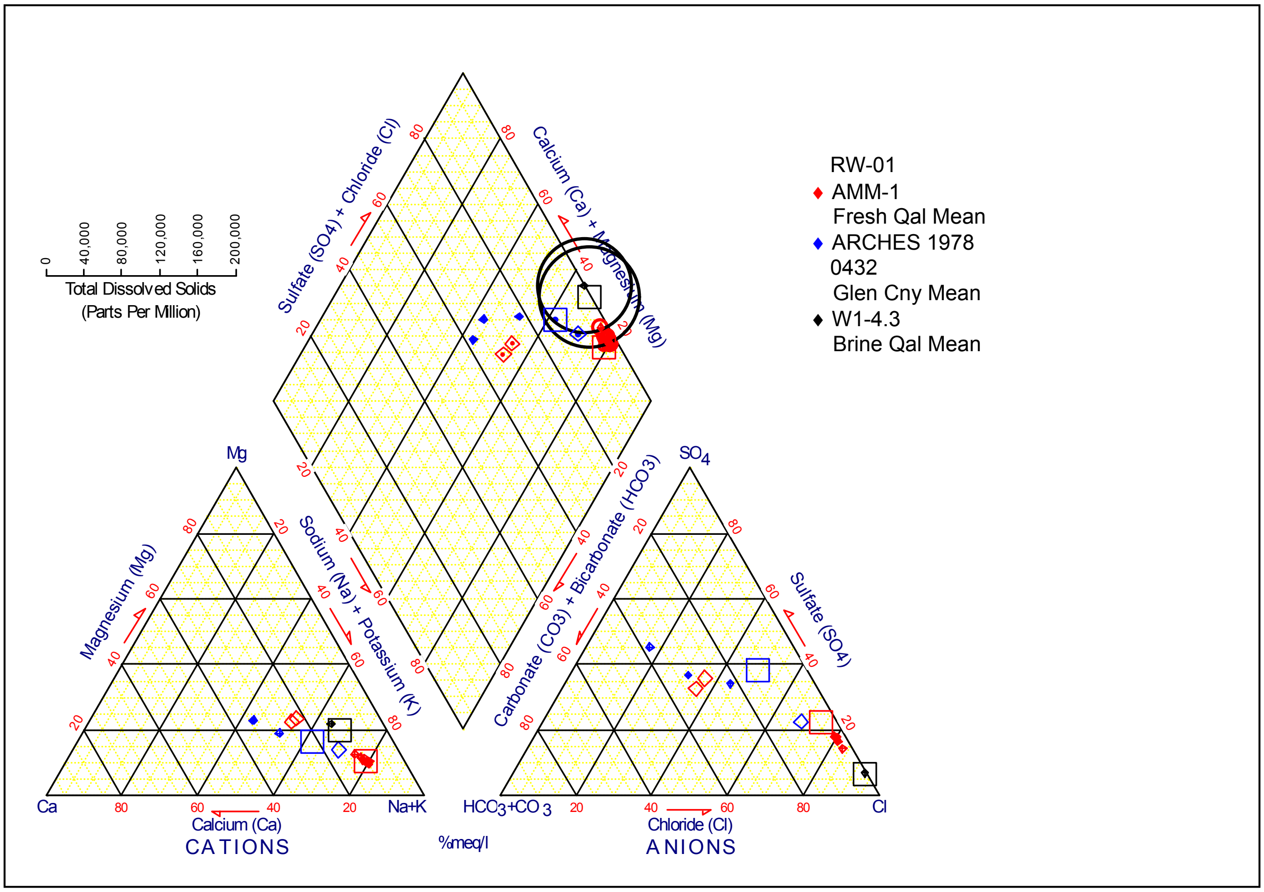


Figure 5–22. Piper Diagram of Background Water Chemistry

Background monitor well MOA-434 located northeast of the site is completed in the Chinle Formation. Limited sampling from this location indicates that the water is very saline, as evidenced by an average TDS concentration of 27,700 mg/L (Table 5–8). Chloride and sulfate concentrations are high, averaging 15,750 and 1,280 mg/L, respectively. Uranium concentrations are detected up to 0.017 mg/L, which is less than the 0.044 mg/L EPA standard in 40 CFR 192. Ammonia concentrations ranging from 0.08 to 0.2 mg/L are low. Trace metals concentrations, including arsenic, barium, cadmium, chromium, lead, molybdenum, selenium, and silver, are all less than their respective standards.

The unconsolidated basin-fill aquifer (Qac and Qaf, see Plates 2 through 11) underlies most of the site. However, only a small portion of the aquifer extends upgradient and beyond the boundary of the former millsite. Two alluvial monitor wells (AMM-1 and MOA-456) are in this area, where the water quality in the aquifer is unaffected by site operations and can be tested. Analytical results of ground water samples collected at AMM-1 and MOA-456 are presented in Table 5–8.

Alluvial water quality at AMM-1 and MOA-456 is characterized by average TDS concentrations of 7,113 and 5,530 mg/L, respectively, which classifies the water as moderately saline. Water quality results presented in the Piper diagram (Figure 5–22) for AMM-1 indicate the dominant anion species is chloride, and the dominant cation is sodium. The sodium-chloride composition probably reflects a small component of the Moenkopi and Chinle Formations, respectively, that are present as subcrops where these wells are located. A small upward hydraulic gradient (see

Section 5.2) measured from the Chinle Formation (MOA-434) to the basin-fill aquifer (MOA-456) would contribute the salinity from the bedrock formations to the basin-fill aquifer.

5.3.4.4 Water Quality Influence from the Paradox Bedrock Formation

The Paradox Formation is not present north of the site; therefore, it is not possible to characterize water quality in the basin-fill aquifer that is influenced by the Paradox Formation directly at an upgradient background location. However, the influence on the water quality can be inferred by examining water quality results from cross-gradient wells installed in the Matheson Wetlands Preserve located across the Colorado River from the site. As discussed in Section 5.3.1, sampling by Cooper and Severn (1994) of shallow ground water wells completed in the basin-fill deposits near the Colorado River indicate relatively high concentrations of TDS and slightly elevated concentrations of arsenic, cadmium, lead, and molybdenum that can be attributed to the underlying Paradox Formation.

DOE conducted additional sampling in the Matheson Wetlands Preserve in December 2002 and March 2003. Locations M11-14, N7-10, and W1-7 were sampled in December 2002 and locations M11-4.8, M11-7, M11-12, M11-14, N7-7, N7-10, N7-11, W1-4.3, W1-7, and W1-10 were sampled in March 2003. These monitor wells are also located near the Colorado River. Results from the sampling are summarized in Table 5−9. The average result is presented for wells that were sampled twice. TDS results are posted in Figure 5–23.

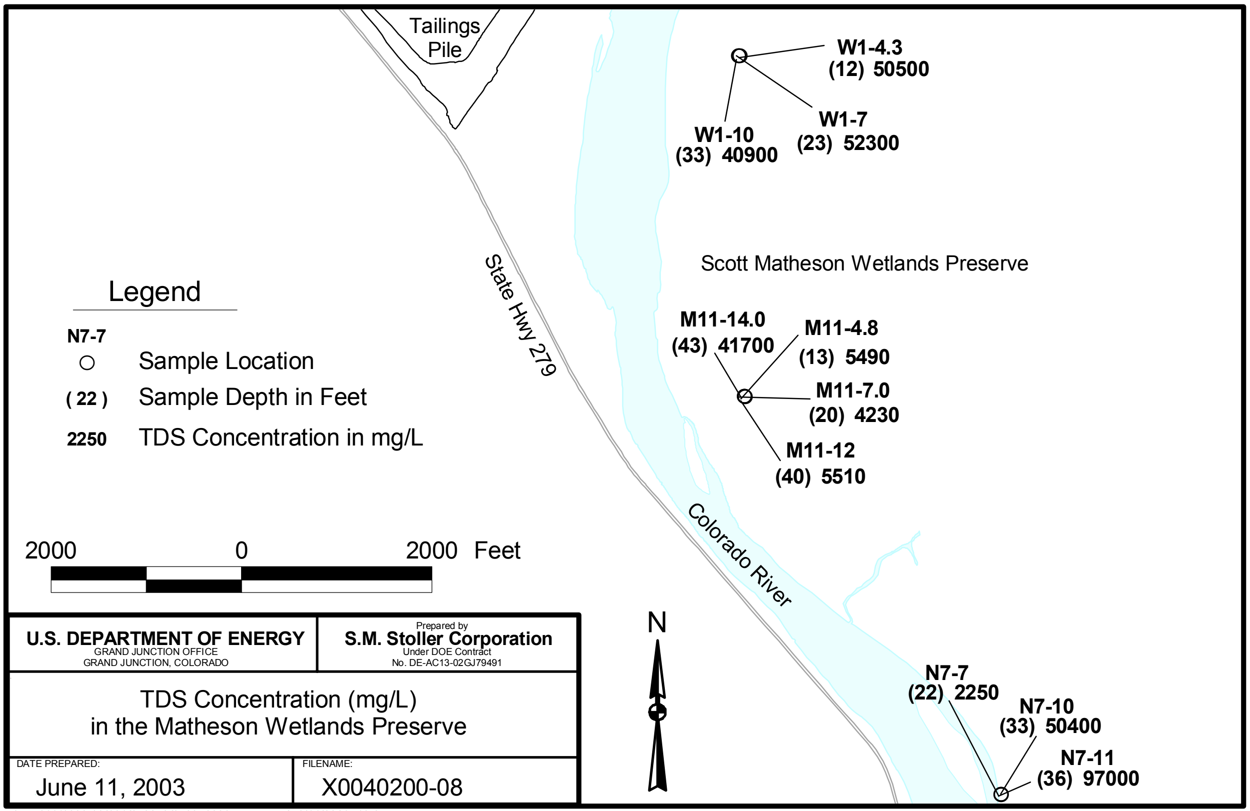


Figure 5–23. TDS Concentrations Measured in the Basin-Fill Alluvium on the Matheson Wetlands Preserve

Table 5–9. DOE Sampling Results from Monitor Wells Completed in the Basin-Fill Aquifer at the Matheson Wetlands Preserve

Analyte	Units	M11-4.8	M11-7.0	M11-12	M11-14.0	N7-7	N7-10	N7-11	W1-4.3	W1-7	W1-10
Major											
Alkalinity, total as $CaCO_3$	mg/L	650	610	448	210	329	165		125	231	
Ammonia, total as N	mg/L	0.46	0.48	0.35	1.55	1.27	1.52	3	0.174	0.257	3
Chloride	mg/L	1,620	1,320	2,550	23,300	905	28,300	52,400	29,700	28,700	23,000
Magnesium	mg/L								2,290		
Nitrate as NO_3	mg/L				0.01		0.075		0.0153	0.0379	
Potassium	mg/L								150		
Sodium	mg/L				16,300		18,000		12,700	12,000	
Sulfate	mg/L	1,440	995	614	2,500	336	2460	5,270	2,940	3,010	2,360
Metals											
Aluminum	mg/L										
Antimony	mg/L								0.00025		
Arsenic	mg/L				0.0046		0.00015		0.0003	0.0005	
Barium	mg/L										
Cadmium	mg/L				0.00005		0.00005		0.0001	0.00005	
Calcium	mg/L								2,370		
Chromium	mg/L								0.0015		
Cobalt	mg/L										
Copper	mg/L								0.007		
Iron	mg/L				17.9		5.17		6.96	22.3	
Lead	mg/L								0.00054		
Lithium	mg/L				0.105		0.12			0.335	
Manganese	mg/L				2.91		0.369		38.5	19.1	
Mercury	mg/L								0.0001		
Molybdenum	mg/L				0.0045		0.0045		0.0045	0.0045	
Nickel	mg/L								0.002		
Selenium	mg/L				0.00005		0.00005		0.00015	0.009	
Silver	mg/L								0.00005		
Strontium	mg/L				24.8		23.2		55.2	65	
Thallium	mg/L								0.00007		
Uranium	mg/L	0.0037	0.0044	0.001	0.001	0.001	0.0033	0.0007	0.0021	0.0231	0.0159
Vanadium	mg/L				0.001		0.001		0.00075	0.0028	
Zinc	mg/L								0.0261		
Other											
Boron	mg/L				1.33		1.11		0.395	0.552	
Dissolved Oxygen (unfiltered)	mg/L	.22	1.48	.08	0.475	6.64	0.2	0.24	1.84	0.28	1.51
Fluoride	mg/L				3.28		3.37			1.85	
Oxidation Reduction Potential	mV	−136	−101	−140	−121	−112	−153	−45	193	−67	−55
Silicon	mg/L										
Total Dissolved Solids	mg/L	5,490	4,230	5,510	41,700	2,250	50,400	97,000	50,500	52,300	40,900
pH (unfiltered)	s.u.	6.98	7.03	7.44	7.05	6.88	7.11	5.7	6.64	6.53	6.27
Physical Measurements											
Density	g/cm^3	1	1	1	1.02	0.999	1.03	1.06	1.04	1.04	1.03
Specific Conductance (unfiltered)	µmhos/cm	7,790	6,140	8,550	57,300	3,380	69,000	111,000	68,600	67,100	59,200
Specific Gravity					1.04		1.04			1.04	
Temperature (unfiltered)	°C	17.1	15.2	14	14.2	8.78	11.4	8.73	14.2	15.9	11.7
Turbidity (unfiltered)	NTU	9.28	2.19	17.7	4.99	105	8.54	95.6	48.3	2.03	373
Radionuclides											
Gross Alpha	pCi/L				180		237			178	
Gross Beta	pCi/L				150		222			150	
Radium-226	pCi/L				0.16		9.26			.145	
Radium-228	pCi/L				6.09		2.6			5.21	
Thorium-230	pCi/L								.8		

Sampling results indicate that TDS concentrations generally increase with depth and proximity to the Colorado River. The interface between the upper fresh water and the underlying brine is sharp, as evidenced by the abrupt change in TDS concentration over a 3-ft interval at locations M11-12 and M11-14. A similar abrupt change in TDS concentrations is observed over an 11-ft interval at locations N7-7 and N7-10. The brine is characterized by TDS concentrations ranging up to 97,000 mg/L, which reflects the influence from the Paradox Formation. Elevated concentrations of sodium, potassium, chloride, sulfate, iron, and manganese are evident, which is consistent with Cooper and Severn (1994) findings. DOE also analyzed the water samples for total ammonia-N and uranium. The highest ammonia concentrations, averaging 3 mg/L, were detected in water samples obtained from well N7-11. Slightly elevated ammonia concentrations ranging from 1.5 to 3 mg/L were detected at M11-14, N7-10, N7-11, and W1-10. These elevated ammonia concentrations are associated with brine samples and probably reflect a minor contribution of the naturally occurring ammonia in the Paradox brines, as reported by Mayhew and Heylman (1965).

The maximum uranium concentration of 0.02 mg/L was detected at location W1-7; this value is less than the 40 CFR 192 standard. Sampling from the DOE study was not sufficient to evaluate trace metals concentrations; however, trace metals results from the Cooper and Severn (1994) study are considered representative of natural background water quality for the Paradox-influenced basin-fill aquifer. The naturally high chloride content of the brine would promote aqueous chloride complexing of some of the trace metals such as copper, lead, and zinc, resulting in the elevated concentrations observed by Cooper and Severn (1994) and Mayhew and Heylman (1965).

5.3.5 Background Water Quality in the Basin-Fill Aquifer Prior to Milling Operations

Monitor wells RW–01, AMM–1, and ATP–3 were previously identified as representative of water in the basin-fill aquifer that was unaffected by the milling process (ORNL 1998a). ATP-3 is believed to represent background water quality; however, this well is not used to determine background conditions as part of this evaluation because it is considered an on-site location.

Additional background monitor wells MOA–432, -434, and -456 installed by DOE allow greater distinction between hydrochemical ground water facies present in the basin-fill aquifer than previous investigations. For this evaluation, the basin-fill aquifer underlying the site is divided into three hydrochemical facies to evaluate background conditions prior to milling operations: (1) an upper fresh to moderately saline facies (fresh Qal) that has TDS concentrations ranging up to 10,000 mg/L, (2) an intermediate brackish facies of very saline water (brackish Qal), having TDS concentration between 10,000 and 35,000 mg/L, and (3) a lower briny facies (brine Qal) that is characterized by TDS concentrations greater than 35,000 mg/L. All three hydrochemical facies existed beneath the site prior to milling activities. TDS concentrations that define the three hydochemical facies and the associated USGS water quality classification is presented in Table 5–10.

Table 5–10. Hydrochemical Facies Based on TDS Concentrations in the Basin-Fill Aquifer

TDS (mg/L)	USGS Water Quality Classification	Hydrochemical Facies in the Basin-Fill Aquifer
Less than 1,000	Fresh	Fresh Qal
1,000–3,000	Slightly Saline	
3,000–10,000	Moderately Saline	
10,000–35,000	Very Saline	Brackish Qal
More than 35,000	Briny	Brine Qal

5.3.5.1 Fresh Qal Facies

The extent of the upper fresh Qal facies is evidenced in Figure 5–20 and Figure 5–21 by the TDS concentrations that are less than 10,000 mg/L. The lowest TDS concentrations occur in the unconfined aquifer in Moab Wash (Qmd) situated at the northwest upgradient boundary of the site and near the Moab Fault zone where inflow from the Glen Canyon aquifer is believed to occur. Background water quality in this area of the site is a sodium−mixed-anion type represented by water samples collected from monitor well RW-01 (Figure 5–23). This fresh water quickly becomes mixed with more saline water in the basin-fill aquifer (Qaf and Qac) as it enters the site and flows underneath the tailings pile toward the Colorado River. Salinity increases with depth and distance from the freshwater source contribution from the Glen Canyon aquifer.

To the east of Moab Wash, along the northern boundary of the site, only a small extent of the fresh Qal facies is present (Figure 5–20). Samples collected from background monitor wells AMM-1 and MOA-456 are representative of the sodium-chloride background water type that occurs in this area. The moderately saline geochemical signature of the background ground water in this area reflects the greater distance from the freshwater source (Moab Wash area) and the proximity to saline bedrock units. Salinity increases with depth as the bedrock formations are approached, as evidenced in the cross-section beginning at MOA-456 and extending southeast to the river (Figure 5–24).

Analytical results from samples collected at monitor wells RW-01, AMM-1, and MOA-456 are summarized in Table 5–11 and are selected to represent the water quality for the upper fresh Qal facies prior to milling operations. Overall, the fresh Qal facies is characterized by low concentrations of ammonia-N, uranium, and other trace metals, all of which have concentrations that are below the EPA standards listed in Table 2–1. TDS concentrations range from 677 to 7,820 mg/L, which classifies the water quality as fresh to slightly saline. Background alkalinity, as calcium carbonate, ranges from 137 to 189 mg/L; sulfate concentrations range from 180 to 1,140 mg/L; calcium concentrations range from less than 47 to 294 mg/L; and magnesium concentrations range from less than 31 to 188 mg/L.

On average, the water pH is near neutral (7.7), and the redox condition is slightly oxidizing (ORP 186 mV). Dissolved chemical species and saturation indices calculated using the PHREEQC geochemical model (Parkhurst et al. 1980) indicate the fresh Qal water is undersaturated with respect to gypsum, suggesting the ground water has not been in contact with the Paradox Formation.

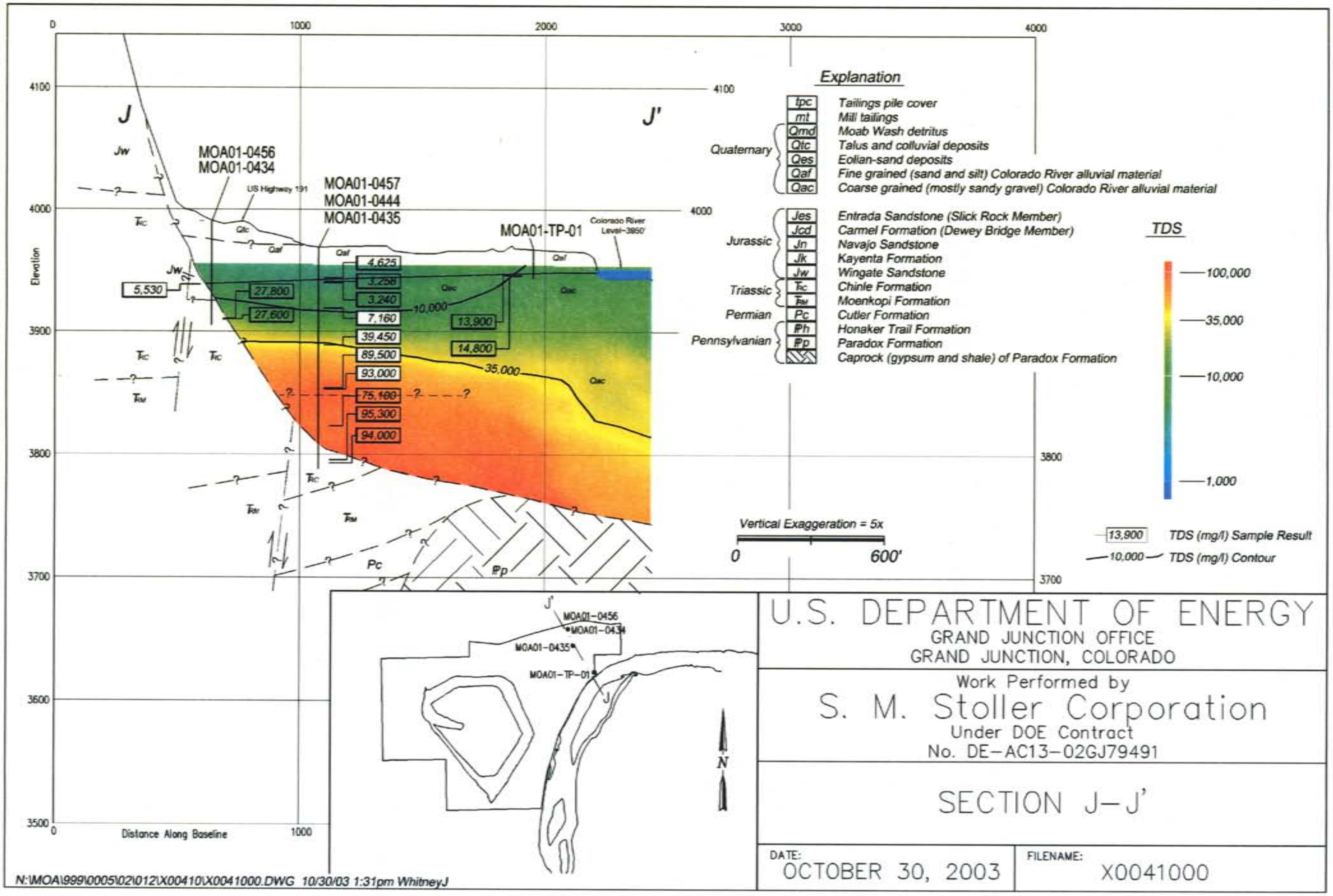


Figure 5–24. Cross-Sectional View of TDS Concentrations in the Basin-Fill Aquifer Beginning at Background Location MOA-456 North of the Former Millsite Area

Table 5–11. Background Water Quality for the Upper Fresh Qal Facies and the Lower Brine Qal Facies Representative of Conditions Prior to Milling Operations

Analyte	Unit	Fresh Qal Facies			Brine Qal Facies		
		Range	Mean	FOD	Range	Mean	FOD
Major							
Alkalinity, total as $CaCO_3$	mg/L	137−189	160	10/10	109−733.04	358	10/10
Ammonia, total as N	mg/L	<0.003−<1	0.0866	1/11	0.0288−3	1.58	10/10
Calcium	mg/L	<47.4−294	135	7/8	1,260−2,370	1650	3/3
Chloride	mg/L	135−4,050	1,990	13/13	17,154−52,388	29,200	12/12
Magnesium	mg/L	<30.7−188	84.5	7/8	1,000−2,290	1,620	3/3
Nitrate as NO_3	mg/L	1.22−15.9	7.58	9/9	<0.02−0.075	0.0346	2/4
Potassium	mg/L	9.29−56	27.7	6/8	59.4−150	92	3/3
Sodium	mg/L	<128−2,080	1,230	11/12	6,370−18,000	12,200	6/6
Sulfate	mg/L	180−1,140	772	13/13	2,004−6,000	3,520	12/12
Metals							
Aluminum	mg/L	<0.0076−<0.051	0.00762	0/4	<0.03−<0.03	0.015	0/2
Antimony	mg/L	<0.0001−<0.011	0.000675	0/7	<0.0005−<0.05	0.0168	0/3
Arsenic	mg/L	0.00018−0.0015	0.000697	8/11	0.00015−0.11	0.0259	4/6
Barium	mg/L	0.0222−0.033	0.028	6/6	0.031−0.121	0.076	2/2
Beryllium	mg/L	0.002−0.002	0.002	1/1	0.0016−0.00165	0.00163	2/2
Cadmium	mg/L	<0.0001−<0.0017	0.000107	0/11	<0.0001−0.014	0.00428	2/6
Chromium	mg/L	<0.0005−<0.011	0.00107	1/7	<0.003−<0.01	0.00383	0/3
Cobalt	mg/L	<0.0013−0.002	0.0011	2/4	0.0451−0.0661	0.0556	2/2
Copper	mg/L	<0.0004−0.005	0.00056	1/7	<0.005−0.007	0.004	1/3
Iron	mg/L	<0.0008−<0.05	0.00928	1/11	0.129−22.3	9.14	6/6
Lead	mg/L	<0.0001−<0.0055	0.00047	0/7	0.00054−0.184	0.113	3/3
Lithium	mg/L	0.0278−1	0.0873	6/7	0.0579−0.335	0.143	5/5
Manganese	mg/L	<0.0001−0.0157	0.00575	1/12	0.369−38.5	11.7	6/6
Mercury	mg/L	<0.0001−<0.0002	0.000095	0/5	<0.0002−<0.0002	0.0001	0/1
Molybdenum	mg/L	<0.0018−0.01	0.00377	4/12	<0.004−<0.009	0.00367	0/6
Nickel	mg/L	<0.0006−0.015	0.00222	1/5	<0.004−0.0647	0.0327	2/3
Selenium	mg/L	0.0091−0.0266	0.0164	11/11	<0.0001−0.009	0.00171	1/6
Silver	mg/L	<0.0001−<0.0055	0.00041	0/7	<0.0001−<0.004	0.00135	0/3
Strontium	mg/L	<2.25−8.45	3.79	10/11	23.2−65	36.8	6/6
Thallium	mg/L	<0.0001−<0.011	0.0007	0/7	<0.00014−<0.00014	0.00007	0/1
Uranium	mg/L	0.0042−0.0259	0.0127	13/13	0.0007−0.0269	0.00768	10/10
Vanadium	mg/L	0.00061−0.0164	0.00534	7/12	<0.0015−0.135	0.0418	3/6
Zinc	mg/L	<0.0006−0.011	0.00231	3/7	0.0179−0.255	0.0812	4/4
Other							
Boron	mg/L	0.106−0.477	0.287	8/8	0.181−1.33	0.685	6/6
Dissolved Oxygen (unfiltered)	mg/L	0.9−6.94	3.65	7/7	0.13−1.84	0.758	9/9
Fluoride	mg/L	<0.193−1.46	0.537	5/7	1.85−3.37	2.83	3/3
ORP (unfiltered)	mV	66.9−619	186	11/11	−184−206	−41.3	10/10
Silica	mg/L	14.3−14.3	14.3	1/1	0.00944−0.0106	0.01	2/2
Total Dissolved Solids	mg/L	677−7,820	4,450	12/12	31,214−97,014	51,400	12/12
pH (unfiltered)	s.u.	7.19−8.11	7.67	14/14	5.7−7.18	6.55	10/10
Physical Measurements							
Density	g/cm3	1.0002−1.0002	1	1/1	1.0195−1.0645	1.04	6/6
Specific Conductance (unfiltered)	µmhos/cm	1,078−14,040	6,920	13/13	43,695−111,287	72,000	10/10
Specific Gravity		1−1.01	1.01	4/4	1.04−1.04	1.04	3/3
Temperature (unfiltered)	°C	14.3−19.4	18.1	12/12	8.73−16.8	12.7	10/10
Turbidity (unfiltered)	NTU	0.2−144	51.4	10/10	0.82−373	88.7	9/9
Radionuclides							
Gross Alpha	pCi/L	<6.73−<73.92	17.6	1/10	<356.33−<473.08	198	0/3
Gross Beta	pCi/L	7.39−64.4	29.2	5/10	<299.19−<444.99	174	0/3
Radium-226	pCi/L	0.07−0.16	0.0633	2/10	<0.29−9.26	3.19	1/3
Radium-228	pCi/L	<0.5−1	0.383	1/10	2.6−6.09	4.63	3/3
Thorium-230	pCi/L	<1.5−<1.6	0.781	0/6	<1.6−<1.6	0.8	0/1

< less than detection limit
FOD - frequency of detection

5.3.5.2 Brackish Qal Facies

The intermediate brackish Qal facies is characterized by TDS concentrations ranging from 10,000 to 35,000 mg/L, which results from a natural mixing of the upper fresh water with the lower brine zone. There are no upgradient monitor wells completed in this facies that can be used to characterize the water quality. Therefore the water quality characterized from the two end points, the upper fresh Qal and the lower brine Qal, bounds the water quality for the intermediate brackish Qal facies.

5.3.5.3 Brine Qal Facies

The lower brine Qal facies is characterized by TDS concentrations greater than 35,000 mg/L that result from the dissolution of gypsum and salt beds in the underlying bedrock formations. Dissolved chemical species and saturation indices calculated using the PHREEQC geochemical model (Parkhurst 1995) indicate the brine Qal water is oversaturated with respect to gypsum, suggesting that gypsum is present along the flow path of this ground water. Most likely the brine Qal derived its chemistry by contact with the caprock of the Paradox Formation, which contains abundant gypsum beds.

Ground water quality determined from water samples collected at selected monitor wells located in the basin-fill aquifer on the Matheson Wetlands Preserve, which is unlikely to have been affected by the milling process, is considered representative of background conditions in the brine Qal facies. Analytical results from the DOE sampling events in 2002 and 2003 (wells M11-14, N7-10, N7-11, W1-4.3, W1-7, W1-10) and from the 1994 Cooper and Severn study (wells M9 and M10) are summarized in Table 5–11 and are selected to represent the water quality from the lower brine Qal facies prior to milling operations. TDS concentrations from two locations are less than 35,000 mg/L but are considered representative of the brine Qal facies. The first sample result of 31,210 mg/L TDS was measured at location M11-14. However, a TDS concentration of 52,200 mg/L was detected in a previous sample from the same location. A sample from the second location (M9) was reported with a TDS concentration of 33,900 mg/L (Severn and Cooper 1994). This sample result is only slightly less than 35,000 mg/L, and therefore is considered representative of the brine Qal facies.

Overall, the background water quality in the brine Qal is poor (Table 5–11). The water is a sodium-chloride type (Figure 5–22) with TDS concentrations ranging up to 97,000 mg/L, which classifies the water quality as briny. Maximum detected concentrations of arsenic (0.11 mg/L), cadmium (0.014 mg/L), and lead (0.18 mg/L) are all slightly higher than EPA standards. Maximum concentrations of uranium (0.027 mg/L) are less than the EPA standard. Secondary drinking water standards are exceeded for sulfate (250 mg/L), chloride (250 mg/L), manganese (0.05 mg/L), and iron (0.3 mg/L), demonstrating the poor quality of the ground water.

5.4 Geochemistry of Tailings Pile Source Area

5.4.1 Background

The mill tailings pile on the Moab site represents the major surficial source of ground water contamination. As described in Section 5.2, an estimated 20 gpm of contaminated pore fluids seeps out of the base of the pile and into the ground water system. The pile contains approximately 10.5 million tons of tailings (NRC 1999); with an estimated average concentration of 0.013 percent uranium in those tailings (SRK 2000), a potential 1,400 tons of uranium is still tied up in the tailings solids, which could eventually leach to the ground water system.

The tailings pile ranges in thickness from about 30 to almost 100 ft; tailings are thickest at the toe of the pile near the river where the initial pond embankment was constructed. The tailings thin in the upslope direction toward the northwest. Tailings were slurried into the tailings pond via a series of pipes; water in the pond had to maintain a distance of at least 250 ft from the surrounding embankments (see Section 3.1). Because the tailings slurry flowed in from the perimeter of the piles, coarser-grained material settled out closer to the embankments, and fines were transported to the center of the pile with the fluids.

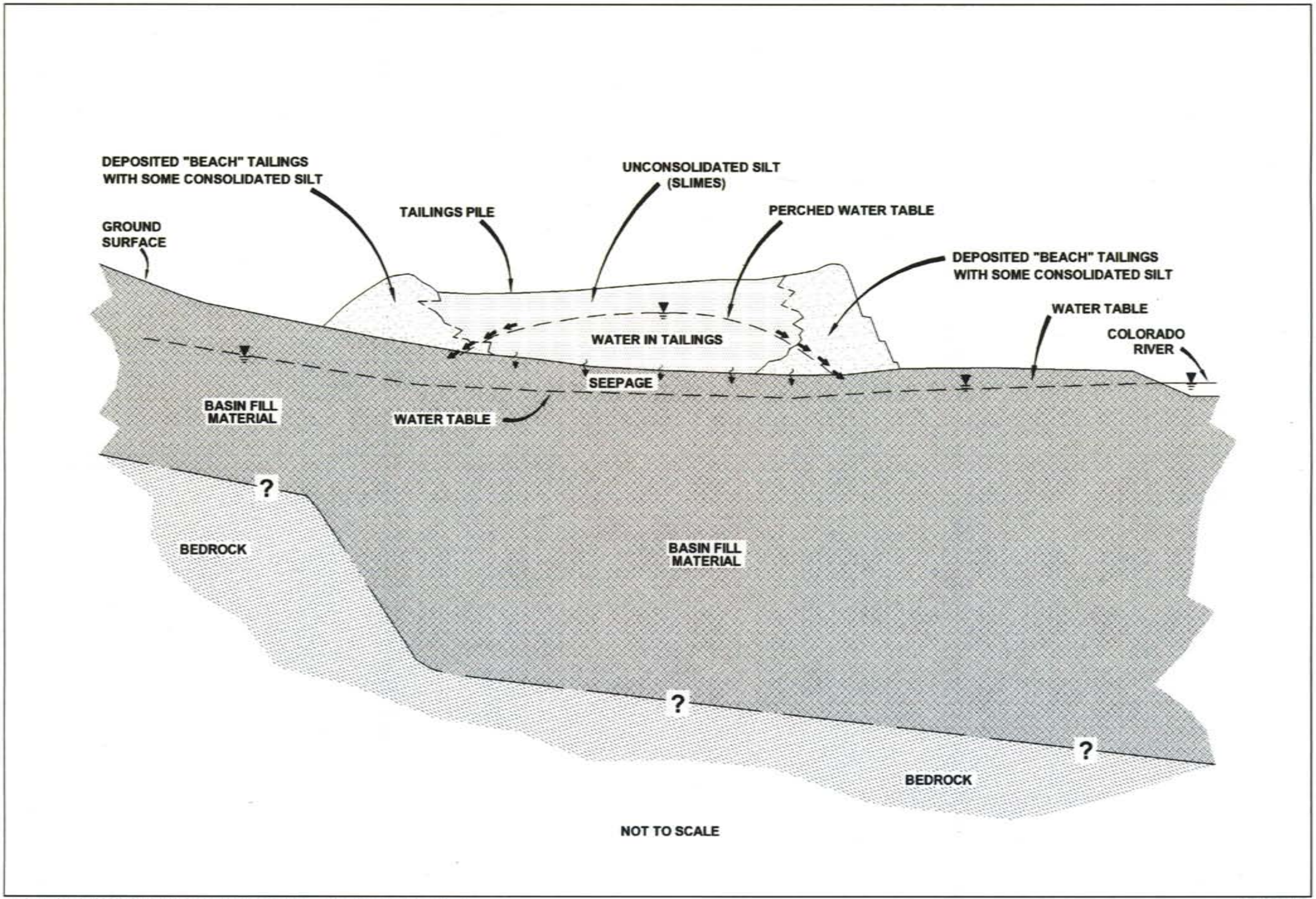
A conceptual depiction of the pile in cross section is shown in Figure 5–25. The bulk of the fluids remaining in the pile reside in the unconsolidated silts ("slimes") in the central portion of the pile. The outer ring of tailings materials have largely been drained of excess water and are generally dry. Some of the fluids seep through the slimes and the very low permeability layer underlying them at a very slow rate. Other fluids migrate laterally from the slimes until reaching the more permeable "beach sand" zones through which they flow to the underlying alluvial material, eventually reaching the alluvial aquifer. Some silty zones are interspersed through the predominantly sandy areas, and some sands are found in the predominantly silty zones. As noted in Section 3.4.1, materials other than mill tailings and process fluids have been disposed of in the tailings pile.

During the milling history of the site, two different processing circuits operated—an acid circuit and an alkaline circuit. The acid circuit operated alone only for the earliest and latest years of site operation; the alkaline circuit was used throughout most of the operating period of the site with or without the acid circuit. The milling history is summarized in Table 5–12 (SRK 2000).

Table 5–12. Milling History

1956–1959	Acid circuit only.
1959–1965	Alkaline circuit only.
1965–1968	Acid and alkaline circuits.
1968–1975	1968: Acid circuit burns down. Alkaline circuit only.
1975–1982	1975: New acid circuit built; both acid and alkaline circuits run.
1982	Alkaline circuit shut down.
1984	Mill placed on standby.

Chemicals used for recovery of uranium in the acid circuit included sulfuric acid, sodium chlorate, iron powder, sodium carbonate, and "conventional solvent extraction" (Atlas 1973). Vanadium recovery in the acid circuit also required the use of ammonia. In the alkaline circuit, uranium processing used sodium carbonate, sodium hydroxide, sulfuric acid, hydrogen peroxide, anhydrous ammonia, and solvent. Recovery of copper in this circuit also included a frothing agent, manganese dioxide, iron powder, and sodium chlorate.



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X0049500

Figure 5–25. Conceptual Cross Section of Tailings Pile Showing the Inner Slimes Zone and the Outer "Beach Sand" Zone

Uranium mill tailings contain the waste residuals from the milling operation. Solid grains in the tailings are composed mostly of quartz, feldspar, and clay minerals resulting from the grinding of the ores. The ores were mostly sandstones and siltstones and contain some carbonate minerals. The color of the bulk tailings is dominated by brown or red-brown, but a substantial portion is gray and gray-green.

Processes occurring in the tailings pond, especially evaporation, caused precipitation of additional minerals such as gypsum ($CaSO_4 \cdot 2H_2O$) and thenardite (Na_2SO_4). Some of these minerals contain contaminants. Contaminants can also adsorb on the surfaces mineral grains in the tailings. Contaminants that do not precipitate or adsorb (or which are later dissolved or desorbed) are retained in the liquid phase, which upon burial becomes pore fluid. The chemical composition of the Atlas tailings and pore fluids has been analyzed (SRK 2000, 2001, ORNL 1998a, SEEPro database), and x-ray diffraction methods have been used to identify tailings minerals (SMI 2001). Samples of tailings pond water were analyzed in 1987 (NRC 1999), and samples from a sump used to store dewatering fluids collected by wick-drains were analyzed by DOE in 2003. Sections 5.4.1.1 and 5.4.2 describe the geochemistry of the liquid and solid phase pile constituents, respectively and report results of studies conducted to evaluate their potential for further mobilization and migration to the ground water system.

5.4.1.1 Tailings Pore Fluids

Dissolved Constituents

In 1987, as part of an independent assessment of the characteristics of the tailings, NRC obtained two samples of the tailings liquid present in the pond at the top of the embankment to identify hazardous organic and inorganic constituents (NRC 1999). Of the 132 organic constituents analyzed, most had concentrations of 0.01 mg/L or less, and all had concentrations less than 0.051 mg/L. The two samples analyzed by NRC (1987) had pH values of 2.17 and 2.19, which are lower than values that have been measured in tailings pore fluids from the site (Table 5–13).

The low-pH samples are apparently representative of the effluent from the acid mill circuit, which was in operation prior to placing the mill on standy. Aluminum, Cr, Cu, Fe, Sb, SiO_2, V, and Zn are enriched in the pond (NRC 1987) samples compared to the tailings pore water samples; most other constituent concentrations in the pond are within the range of pore water concentrations (Table 5–13). The “facies” designation in Table 5–13 refers not to the actually lithology of the sample collected, but of the zone from which the sample was collected (inner slimes zone or outer “beach sand” zone; see Figure 5–25).

Analysis of the AR series of samples collected by SRK (2000), the PW series collected by Atlas, and samples 0438, 0439, and 0538 through 0543 collected by DOE indicate that TDS concentrations in tailings pore water range from 8,840 to 240,000 mg/L, with a mean[1] of 35,200 mg/L; the mean pH is 6.63 (Table 5–13). In order of descending concentration, the dissolved solids in the tailings pore water are dominated by SO_4, Na, Cl, alkalinity (total as $CaCO_3$), NH_3-N, Mg, and Ca. These constituents result mainly from ore-processing fluids. The

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[1] Geometric means are used instead of arithmetic means throughout this section because many geochemical distributions are log-normal. However, no statistical tests were applied to this data set to confirm that data are lognormally distributed. Depending on the analyte, alternative distributions could be more applicable.

mean concentrations of uranium and ammonia in the pore fluid samples are 6.7 and 1,607 mg/L, respectively (Table 5–13). Though most pore fluid samples have elevated ammonia levels, several samples had levels that could be considered low (2 orders of magnitude less than the average). The samples may be more representative of fluids generated by the acid leach circuit, which only used ammonia for extraction of vanadium-bearing ores. The fact that the majority of samples have much higher concentrations of ammonia probably reflects the dominance of the alkaline circuit operation at the site. Sample locations are shown in Figure 5–26.

About 17,000 vertical band drains ("wick drains") were installed from September through December 2000 to dewater the tailings (Henderson et al. 2002). Pore water flows up the wick drains and horizontally to a collection sump. DOE sampled the sump in September 2002, December 2002, and October 2003. The most complete analysis was conducted in September 2002, and these results are included in Table 5–13. The TDS concentration in the sump sample is 216,000 mg/L, about an order of magnitude more than the mean pore water concentration. The sample contains much higher concentrations of major ions than any of the pore water samples, for example: 185,000 mgL SO_4, 38,300 mg/L Na, 16,411 mg/L NH_3-N, 6,300 mg/L Cl, and 5,610 mg/L Mg (Table 5–13). Dissolved Fe and Mn concentrations of 933 and 218 mg/L, respectively, are much higher than in the pore water. Base metals such as Cu, Pb, and Zn and many other constituents are also concentrated in the sump water. In contrast, U and Mo concentrations in the sump are nearly the same as in the pore water. The origin of the elevated concentrations of major ions and contaminants in the sump is believed to be dissolution of salt deposits that were precipitated in the upper portion of the tailings. The pore fluid moving up through the wick drains is predominantly from the upper portion of the tailings. As this pore fluid slowly moves up the wicks, it dissolves salts in the upper portion of the tailings and is then directed horizontally through wicks to the collection sump. The sump sample location (0537) is shown in Figure 5–26.

Three samples of pore fluids have TDS concentrations similar to the TDS concentration measured in the sump. These three samples were collected from shallow (9-ft bgs) lysimeters in the central portion (slimes facies) of the tailings (Figure 5–26). There seems to be a relationship between depth and TDS; shallower samples have higher TDS (Table 5–13). SRK (2000) also observed a decrease in TDS with depth based on specific conductance measurements made on paste samples of tailings. The high TDS strata near the surface are also low in pH.

The shallow, low pH, high TDS, high NH_3-N zone is likely caused by the change to acid milling during the last 4 years of mill operation. It also reflects a longer period of evaporation of the residual mill pond following mill closure. Though the acid circuit did not use as much ammonia as the alkaline circuit, evaporation has probably concentrated whatever ammonia was present. Such a concentrating process is supported by concentrations of some metals in the 1987 pond and more recent sump samples. Aluminum, copper, iron, vanadium, and zinc are all much higher in the pond/sump samples than any of the pore fluid analyses. Increased concentrations of pore fluids in the upper layer could also be due to pore water extraction with surface evaporation conducted from 1990 to 1996.

Table 5–13. Chemical Composition of Tailings Pond Water, Tailings Pore Water, and Tailings Sump Samples

	Units	AR11S	AR2S	AR13S	AR9S	AR12S	PW-1	PW-2	PW-6	PW-9	PW-11	PW-04 OB-A	0438	0439	0538	0544	0539	0540	0541	0542	0543	AR8S	0545	0546	Pore Water	NRC 1	NRC 2	Sump (0537)	Sump (0537)
Facies		**Sand**															**Slimes**								**All**	**Pond**		**Drains**	
Sample Type		**sand**	**slime**		**slime**	**mixed**	**mixed**	**mixed**	**mixed**	**mixed**	**mixed**	**mixed**	**slimes**	**slimes**	**na**	**na**	**na**	**na**	**na**	**na**	**na**	**slime**	**na**	**na**	**Gm. Mean**				
Sample Date		**5/7/00**	**5/2/00**	**5/10/00**	**5/5/00**	**5/6/00**	**11/25/97**	**11/25/97**	**11/24/97**	**11/24/97**	**5/30/02**	**5/30/02**	**8/20/02**	**8/8/02**	**4/3/03**	**4/3/03**	**3/28/03**	**3/28/03**	**3/28/03**	**3/28/03**	**3/28/03**	**5/8/00**	**4/3/03**	**4/3/03**		**1987**	**1987**	**9/19/02**	**3/26/03**
Depth (ft)		**48-68**	**24-44**	**24-44**	**64-84**	**29-49**	**49-89**	**36-76**	**4-74**	**9-69**	**5-65**	**8-38**	**44**	**27**	**14**	**15**	**9**	**9**	**9**	**9**	**9**	**28-47**	**30**	**30**					**9**
Pb-210	pCi/L	57.6	24	59.2	167.3	31.9					19.04	32.61		15.29								22.6			35.68			199.23	
Po-210	pCi/L	<3	<3	35.2	6.6	<3					3.49	7.27		3.76								13.9			5.87			41.4	
Ra-226	pCi/L	52.6	282.8	33.8	32.3	48.8					52.73	39.23		67.3								107.4			61.09			125.4	
Ra-228	pCi/L	1.5	1.6	<1	<1	<1					4.06	5.21		<2.92								1.9			1.87			11.19	
Th-230	pCi/L	3.1	2.3	1.0	7.2	1.0					<8	<1.6		6.5								13.5			3.38			<7.4	
U-234	pCi/L										12,700	3,490		4,290											5,750.37				
U-238	pCi/L										12,600	4,000	1,440	5,310											4,430.70				
Ag	mg/L	<0.0005	<0.0010	<0.0005	<0.0005	0.002					0.0019	0.0013										<0.0005			0.0009	<0.5	<0.5	0.0014	
Al	mg/L	0.075	3.27	0.334	0.046	0.075							<0.038	0.0705								37.7			0.27	200	450	77.2	
Alkalinity	mg/L	1,075	1,506	834	1,346	2,104	1,324	1,932	2,360	1,934												77			1,131.4	<5	<5	1,300	
Gross Alpha	pCi/l	6,012.4	1,127.8	1,614.7	10,832.6	14,707.6					16,473	5,783	3,163									1,733.7			4,593.9			8,456	
As	mg/L	1.51	0.961	0.066	0.412	0.362					0.0099	0.133	0.0481	0.151								0.276			0.1810	1	1.8	0.0045	
B	mg/L	0.16	0.42	0.33	0.18	<0.1							0.127	0.0606								0.8			0.20	<0.8	<0.8	1.53	
Ba	mg/L	0.069	0.032	0.055	0.04	0.047					0.0171	0.0218										0.064			0.039	<0.2	0.25	0.0653	
Be	mg/L	<0.001	<0.002	<0.001	<0.001	<0.002							<0.0005	<0.0005								0.01			0.0013	0.062	0.14		
Gross Beta	pCi/l	6,429.4	1,220.7	1,524.0	11,471.0	14,414.0					11,383	4,020	862									1,681.7			3,703.9			4,069	
Bi	mg/L	<0.002	<0.004	<0.002	<0.002	<0.004																<0.002			0.0025				
Ca	mg/L	43.3	432	318	16.6	176					452	450	214	507								383			207.3	130	310	652	
Cd	mg/L	<0.002	0.764	0.012	0.004	<0.003					0.0483	0.0304	0.0064	0.0492								1.09			0.025	0.22	0.49	1.98	
Cl	mg/L	1,443.7	798.9	1,597.7	1,639.7	830.8	915	550	733	612	736	638	951	906	768	299	5,409	6,480	5,795	1,144	1,213	1,681.3	1,202	206	1,082.4	410	370	6,300	5,584
Co	mg/L	0.002	0.957	0.112	0.001	0.176							0.0074	0.294								0.974			0.0547	0.61	1.3	6.96	
Cr	mg/L	0.004	0.007	0.004	0.004	0.007					<0.0006	<0.0006	<0.0025	<0.0025								0.021			0.0033	0.7	1.3	<0.003	
Cu	mg/L	0.014	0.047	0.038	0.024	0.162					1.21	0.409	0.012	0.026								4.14			0.095	5.1	11	12.7	
Density	g/mL														1.0645	1.0122	1.1681	1.1885	1.1877	1.0105	1.0395		1.0073	1.0475	1.078				1.1193
DO	mg/L	4.57	3.45	2.34	6.7	3.5									2.16		1.5	1.36	1.57	2.1	1.58	0.82	1.14	1.63	2.1				
F	mg/L	<10	<10	25.9	<10	<10							5.16	11.8								43.5			12.72	<100	<100	56.2	
Fe	mg/L	0.06	48.5	14.3	<0.05	<0.05					0.0954	0.01	<0.02	<0.02								211			0.311	270	650	933	
Hg	mg/L	0.0006	0.001	0.0006	0.0005	0.001					<0.0002	<0.0002										0.0025			0.0006	<0.0005	<0.0005		
K	mg/L	22.8	218	36.9	20.4	300					281	334	50.7	327								128			108.4			648	
Li	mg/L	<1	2.41	<1	<1	1.2							0.126	0.629								1.9			0.901	<2	3.7	8.31	
Mg	mg/L	22.9	995	119	16.3	1,040					1,190	1,690	93.7	1,620								459			302.6	230	500	5,610	
Mn	mg/L	0.044	64.8	4.61	0.074	3.52					7.95	16.1	0.0106	52.6								39.7			2.50	13	28	218	
Mo	mg/L	8.7	5.03	8.96	10.3	1.95					1.48	1.32	2.78	2.54								1.37			3.29	<0.4	0.52	1.01	
Na	mg/L	2,830	6,010	2,840	2,590	6,070					6,260	5,230	4,230	8,790								5,380			4,673.8	800	1,800	38,300	
NH_3-N	mg/L	40.3	3,430	74.9	38.2	2,390	1,070	2,470	3,940	1,800	3,936	4,356	88	5,320	7,500	1,000	19,000	21,500	21,500	950	3,300	2,510	200	3,700	1,606.7	1,672	1,867	16,411	13,500
Ni	mg/L	0.007	0.409	0.047	0.008	0.299					0.299	0.596	0.0153	0.165								0.816			0.10	<0.6	1.1		
NO_2-N	mg/L																									<30	<30		
NO_3-N	mg/L														347	54	1,987	1,620	1,316	<282	273		13	366	313.3				1,521
NO_3+NO_2-N	mg/L	0.23	65.1	1.73	<0.2	13.5					85	77	1.39	245								124			10.68	<112	<112	1,533	
ORP	mV														100		168	76	131	-47	69		-121	171	68.375				
P	mg/L	2.83	5.06	0.77	0.83	1.43							0.28	1.41								1.46			1.29	<3	5.1	<0.88	
Pb	mg/L	0.001	0.001	0.0007	0.0007	<0.0010					<0.0001	0.00057	<0.0005	0.0025								0.0005			0.0007	<2	<2	0.0609	
pH	s.u.	8.6	6.43	6.7	8.6	7.6	6.96	9.8	6.55	7.2			7.94	6.77	6.42		4.32	4.75	4.67	6.36	5.65	5.9	6.73	4.69	6.63	2.17	2.19	5.92	
Sb	mg/L	0.008	0.003	0.002	0.007	<0.002					0.00082	0.0032	0.0057	0.0026								<0.001			0.0028	11	24	0.0016	
Se	mg/L	0.02	0.696	<0.005	0.017	0.141					0.235	0.813	0.0116	1.16								0.375			0.1049	0.23	0.45	4.49	
SiO_2	mg/L	7.5	31	13.4	6.2	7.6							6.9	33.6								9.4			11.6	24	51		
Sp. Cond.	µS/cm	7,040	12,640	7,460	6,940	13,290																11,930			9,489.9				
SO_4	mg/L	3,561.8	26,321.8	4,772.2	2,726.4	24,614.2	15,790	23,680	31,480	18,500	30,100	31,800	8,430	39,300	62,274	11,208	159,011	178,149	187,912	7,347	33,760	20,280.6	5,237	40,076	22,076.1	26,000	30,000	185,000	114,555
Sr	mg/L										11.5	10.1	7.54	9.81											9.63	1.7	3.6	19.7	
TDS	mg/L	9,510	35,900	10,900	8,840	34,600					43,100	44,000	14,800	57,200	77,049	15,285	238,798	240,031	236,715	11,643	45,412	28,600	10,307	54,299	35,236.7	22,800	23,900	216,000	152,241
TOC	mg/L												9.7	107											32.2				
Tl	mg/L	0.003	0.038	0.006	0.002	0.019					0.0318	0.0169	0.0061	0.0245								0.027			0.012			0.269	
U	mg/L	22.2	2.44	6.95	39.9	55.3	26.5	19.8	21.9	25.7	38.2	11.2	4.19	13.9	1.058	2.027	6.126	3.16	0.653	5.057	0.32	5.25	1.866	3.976	6.7	4	8.9	15.3	14.387
V	mg/L	3.41	3.16	<0.002	2.5	0.098					0.0292	1.53	0.231	2.39								<0.002			0.22	24	53	2.57	
W	mg/L	0.017	<0.003	<0.002	0.025	0.003																<0.002			0.0050				
Zn	mg/L	0.016	1.92	0.046	0.015	0.08					0.19	0.97	0.0048	0.427								3.59			0.137	2.6	5.9	18.2	

NOTE: NRC 1987; SRK 2001; SEEPro; Sump is SEEPro database location 0537. NRC 1 and NRC 2 are samples of the tailings pond collected by NRC in 1987; AR samples are tailings pore fluids collected by SRK in May 2000; Sump is a sample of the wick-drain collection sump collected by GJO on September 9, 2002.

TDS = total dissolved solids, Sp. Cond. = specific conductance; µS/cm = microsiemens per centimeter, ORP = oxidation-reduction potential; ND = not detected, DO = dissolved oxygen; TOC = total organic carbon; “<” indicates less than detection (detection limit used in means)

*Geometric Mean of all analyses

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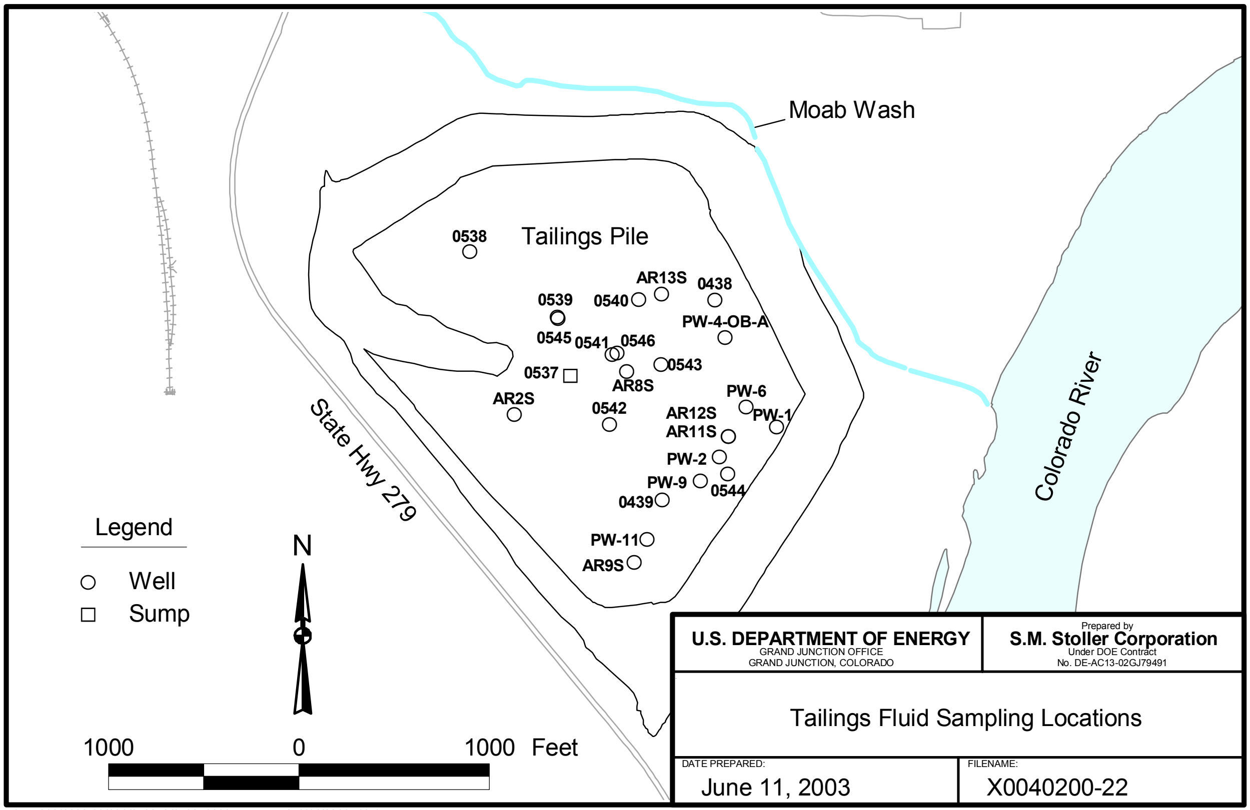

Figure 5–26. Tailings Fluid Sampling Locations

Oxidation-Reduction Potential

The oxidation-reduction potential of the tailings can control the dissolution and precipitation of redox-sensitive minerals and, thus, can affect the composition of pore fluids. The minerals in the tailings were subjected to extreme oxidizing conditions during milling (e.g., addition of strong oxidizers such as H_2SO_4 and sodium chlorate) and will retain an oxidized state unless subsequently exposed to reducing conditions in the tailings pile. Constituents such as sulfide minerals or organic material that can produce reducing conditions are sparse in the tailings. It is unlikely that a robust population of microbial reducer is present due to the absence of carbon sources. Therefore, it is unlikely that conditions in the tailings pile have become anaerobic.

The oxidation-reduction potential of the pore fluids has not been measured, but some other parameters can help to establish the relative oxidation state. Dissolved oxygen concentrations in the pore water samples range from 0.82 to 6.7 mg/L with a mean of 2.1 mg/L, suggesting relatively oxidized conditions (Table 5–13). In a few samples, dissolved Fe and Mn concentrations of up to 211 mg/L and 64.8 mg/L, respectively, suggest reducing conditions; however, some of the high Fe and Mn concentrations are likely caused by lower pH values. The color of the tailings ranges from red brown to gray green, suggesting various oxidation states of Fe.

Mineral Saturation Indices

SRK (2000) computed saturation indices for six tailings pore fluids using the geochemical speciation program MINTEQA2. From this evaluation they conclude that (1) most samples are near saturation with gypsum ($CaSO_4 \cdot 2H_2O$), suggesting a control for SO_4 concentration, (2) Al concentration may be controlled by gibbsite [$Al(OH)_3$] or amorphous aluminum hydroxide [am $Al(OH)_3$], (3) barite ($BaSO_4$) is always oversaturated, (4) U solubility may be controlled by equilibrium with uraninite (UO_2) or schoepite [$UO_2(OH)_2$ H_2O], or it may be controlled by the rates of oxidation, and (5) some ferric oxides and oxyhydroxides are oversaturated under oxidized conditions, suggesting the presence of variably oxidized and reduced conditions in the tailings. Mineral saturation indices in tailings pore fluids can indicate relative trends, but the values should be used with caution because the ionic strengths far exceed the empirical limits of the activity coefficient equations used in the program. Results from column tests (DOE 2003c) confirm that sulfate minerals are controlling pore fluid chemistry in the acid-milled tailings.

5.4.2 Tailings Pile Solids

Most of the by-products of uranium milling at the Moab site are still contained in the tailings pile. Using the average concentration of uranium and the volume of the tailings pile, an estimated 1.3 million kg of uranium still resides in the mill tailings. Containment of this large mass of by-products is an important consideration in protecting human health and the environment. Because a small flux of pore water is released from the tailings and passes into the underlying ground water (Appendix D, Calculation X0023400), there is a potential for chemical by-products to contaminate the ground water system. Solids by-products can contribute to ground water contamination only if they dissolve into the aqueous phase.

5.4.2.1 Mineralogy

In separate studies, SRK (2000) and the University of Wyoming (SMI 2001) examined the mineralogy of different tailings samples using x-ray diffraction and scanning electron microscopy. Quartz is the dominant mineral in all samples, and nearly all samples contain abundant clay minerals. Feldspar (albite and microcline), calcite, muscovite, and dolomite are present in most samples. Sulfate minerals (barite, gypsum) were also identified along with those currently in the iron-, molybdenum-, and vanadium-bearing phases. Clay minerals included illite, smectite, iron chlorite, and kaolinite. SRK reported the presence of lead sulfide (galena) in at least one sample, though no reduced minerals were identified by the University of Wyoming study (SMI 2001).

Minerals that host the contaminants in the tailings are not well known. Determining contaminant residence is not easily accomplished with existing mineralogic methods. SMI (2001) conducted sequential selective extractions of two tailings samples (tailings red and tailings gray) to help determine which minerals contain the contaminants. The idealized concept of selective extractions is that a solvent will selectively extract one specific mineral or mineral group. Removal of a contaminant during a selective extraction provides evidence that the contaminant is associated with the targeted mineral group. Unfortunately, results of selective extractions are difficult to interpret because no solvents are perfectly selective. Despite this shortcoming, selective extractions provide one of the few methods available to evaluate contaminant residence, and the results offer some insights into this issue.

A complete set of data from the sequential selective extraction study is presented in SMI 2001, Appendix B-4; only a summary of the conclusions for the mill tailings is presented here. Little iron was removed during the first three extractions, reflecting the oxidizing conditions of the extractions (Table 5–14). Uranium and sulfate were released in variable proportions by the different extractants, indicating they are tied up in multiple phases; however, more SO_4 is removed prior to the HNO_3 digestion, indicating that more SO_4 is present in water-soluble minerals. Sulfate concentrations were high in the extractant solutions, and solubility limits may have been exceeded in which case, the residence of all the SO_4 in water-soluble minerals such as gypsum or thenardite cannot be ruled out.

Table 5–14. Partial Results From Sequential Extraction Analyses (SMI 2001) (in percent extracted)

Sequential Extractants	Iron (Percent)	Uranium (Percent)	Sulfate (Percent)
Tailings Gray			
1 M $NaHCO_3$	0.00	12.84	18.57
1 M $MgCl_2$ at pH 7	0.00	2.23	30.29
1 M NaOAc at pH 5	0.05	4.15	11.38
1 M $NH_2OH{\cdot}HCl$ in 25% HOAc at pH 2	3.36	2.67	10.09
HNO_3/H_2O_2 digestion	96.59	78.11	29.67
Tailings Red			
1 M $NaHCO_3$	0.01	21.11	71.36
1 M $MgCl_2$ at pH 7	0.00	5.66	8.12
1 M NaOAc at pH 5	0.05	17.26	5.98
1 M $NH_2OH{\cdot}HCl$ in 25% HOAc at pH 2	1.26	27.46	5.44
HNO_3/H_2O_2 digestion	98.69	28.51	9.10

Alpha-track mapping, electron microscopy, selective extraction, and petrography were used to study the mineralogical residence of alpha-emitting contaminants in tailings from three uranium milling sites that used milling processes similar to those used at the Moab site (Morrison and Cahn 1991). These studies concluded that (1) there are distinct differences in contaminant residences between acid- and alkaline-milled tailings, (2) in acid-milled tailings, most of the Ra-226 is contained in barium and strontium sulfate grains, (3) alpha emissions in alkaline-milled tailings come predominantly from composite siliceous grains; siliceous grains form because silica is mobile at the elevated pH values of alkaline milling. Similar residences for alpha-emitting constituents are probably present in the Moab tailings. Contaminants residing in siliceous grains are less likely to be leached into migrating pore water than those residing in more soluble phases.

5.4.2.2 Bulk Chemistry

Atlas drilled four borings into the tailings pond in March 1979, collected samples of tailings solids, and analyzed for U_3O_8 and V_2O_5 (SRK 2000). The mean concentration of U_3O_8, determined from analyses of 66 tailings samples, was 0.013 percent. The mean concentration of V_2O_5, determined from analyses of 58 tailings samples was 0.094 percent. In 1988, Atlas analyzed one slimes tailings sample and one sand tailings sample for Ra-226 (SRK 2000). Radium-226 concentrations were 270 and 890 pCi/g in these two samples. No information was found regarding the digestion methods used for the U_3O_8, V_2O_5, or Ra-226 determinations.

On December 10 and 11, 1997, personnel from Oak Ridge National Laboratory (ORNL) drilled and sampled borings PB-1 and PB-2 on the tailings pile (ORNL 1998a). Different digestion methods were used for the uranium analyses (complete digestion) compared with the other constituents (acid digestion). However, most of the uranium and other analyzed constituents likely reside in the acid-soluble fraction and, thus, the results of the two digestion methods are probably comparable for the use here. Six slimes samples and one sand sample of tailings were analyzed.

Concentrations of As, Ba, Cu, Mo, Se, U, and V are enriched in the ORNL tailings samples compared to mean crustal abundances and mean concentrations in the alluvial aquifer between the river and the tailings (Table 5–15). All of these constituents are present in the raw ores; Ba was also used in the processing to remove Ra-226. Contrary to what was expected, the highest concentrations of most constituents were in the sand sample rather than in the slimes samples.

Table 5–15. Nonradionuclide Chemistry of Tailings Samples From ORNL (1998)

Sample Identity	Sample Description	Arsenic (mg/kg)	Barium[a] (mg/kg)	Copper (mg/kg)	Molybdenum (mg/kg)	Selenium (mg/kg)	Uranium (mg/kg)	Vanadium (mg/kg)
PB-1-83	Slimes Tailings	12.9	343.0	29.4	71.9	0.4B	103.0	525.0
PB-1-85	Slimes Tailings	13.2	206.0	45.5	65.3	0.2U	176.0	649.0
PB-1-89	Slimes Tailings	14.8	267.0	95.6	105.0	0.2U	163.0	1,510.0
PB-2-45	Sand Tailings	113.0	1,290.0	832.0	78.8	3.7	338.0	597.0
PB-2-74	Slimes Tailings	9.2	340.0	21.5	27.6	0.34	65.8	496.0
PB-2-76	Slimes Tailings	7.7	276.0	36.1	68.3	0.3B	231.0	489.0
PB-2-78.5	Slimes Tailings	78.9	365.0	306.0	133.0	1.2	158.0	2,460.0
Mean Crustal Abundance[b]		1.8	426	55	1.5	0.05	1.8	135
Downgradient Aquifer[c]		2.1	87.6	6.2	1.6	0.7	2	14.1

[a] All barium results are flagged, E = reported value is estimated because of the possible presence of interference, and N = spike sample recovery is not within control limits.
[b] From Mason and Moore 1982.
[c] From SMI 2001 (Appendix B-4, Table 1)
Notes: U = detection limit, B = estimated value.

SRK (2000) reports bulk chemical analyses of 38 solid samples collected from the tailings (laboratory reports are included in Appendix B of the SRK report). Arithmetic means for those results are presented in Table 5–16. Mean abundances of constituents in the earth’s crust and mean concentrations of constituents in alluvium downgradient of the Moab site are provided for comparison. The samples were digested using EPA standard method 3050 (EPA 1998), which will not completely digest most samples but will likely remove all of the milling-related constituents. Ammonia was treated separately, being extracted with a strong KCl solution (personal communication with Tracy Delany at Energy Laboratories).

Mean NH_3-N concentrations range from 11.6 mg/kg in the embankment material to 2,821 mg/kg in samples of consolidated silt; the tailings values are considerably higher than downgradient alluvium or average crustal values. The NH_3-N is derived from ammonia used in the milling circuit. Concentrations of As, Cd, Cu, Pb, Mo, Ni, U, and V are also high in tailings (Table 5–16), reflecting the presence of these constituents in the uranium ores. Mean Fe concentrations range from 2,557 to 10,996 mg/kg in the tailings and are not significantly higher than concentrations in downgradient alluvium. The Fe concentrations are less than typical concentrations of Fe in the earth’s crust.

Table 5–16. Arithmetic Means of Nonradiogenic Tailings Constituents From SRK (2001) (downgradient alluvial aquifer data from SMI 2001 [Appendix B4, Table 1])

Parameter No. of Samples	Units	Unconsol. Silt (slimes) 6	Consol. Silt 10	Dry Silt 2	Wet Sand 6	Dry Sand 4	Embankment Material 3	Downgradient Aquifer 16	Mean Crustal Abundance[1]
Paste pH	s.u.	7.7	7.5	-	7.8	4.6	8.4	-	-
Paste TDS	mg/L	6.1	5.3	-	7.1	3.8	0.2	-	-
Ammonia-N	mg/kg	1,715	2,821	-	532	330	11.6	173	20[2]
Arsenic	mg/kg	32.0	46.1	6.84	21.40	11.7	33.5	2.1	1.8
Cadmium	mg/kg	13.6	9.3	2.35	4.54	5.34	3.4	<0.2	0.2
Copper	mg/kg	236	167	23.1	94.6	61.8	84.5	6.2	55
Iron	mg/kg	10,996	9,283	4,211	4,136	2,557	7,473	5,905	50,000
Lead	mg/kg	112	92.2	54.8	47.9	46.7	71.0	5	13
Molybdenum	mg/kg	3.8	26.7	28.4	8.49	13.0	29.1	1.6	1.5
Nickel	mg/kg	41.4	21.2	6.41	9.77	6.14	13.3	6.7	75
Uranium, elemental	mg/kg	137.6	103.5	80.7	17.7	36.18	57.8	2	1.8
Vanadium	mg/kg	1,004	1,124	564	432.6	211	970	14.1	135

[1] From Mason and Moore 1982.
[2] Total nitrogen

Tailings and subpile soil data collected by DOE were presented in Section 4.8 and Appendix D, Calculation X0023400. Ammonia concentrations (NH_3-N) ranged from 202 to 437 mg/Kg and uranium concentrations ranged from 89.2 to 191 mg/Kg. These results are consistent with those of previous studies.

5.4.2.3 Soluble Salts, pH, and Acid-Generating Capacity

SRK (2000) used the following method to determine the pH and specific conductance of 124 samples of tailings: (1) sample was sieved to 5 mm, (2) 35 mL of sieved material was mixed with 100 mL of deionized water, (3) the slurry was allowed to sit for 15 minutes, after which pH and specific conductance were measured with probes. Mean pH values of most of the tailings samples were near neutral but ranged from acidic (4.6) to basic (7.8); individual pH values ranged from 2.9 to 9.9 (SRK 2000). A 3-ft zone of sand tailings at about 5 ft bgs has a pH of about 2.5 to 3. SRK (2000) suggests this low pH zone could have resulted from a change to acid milling. Another low pH zone (pH about 4) exists near the base of the tailings at about 70 ft bgs. Between these two zones the tailings have pH values of about 7 to 10. The specific conductance values on the slurries provide an indication of the amount of soluble salts in the tailings. The specific conductances indicate relatively high soluble salts at the top of the tailings down to about 10 or 15 ft, and lower concentrations from 15 ft bgs to the base of the tailings (SRK 2000). These changes probably reflect the changes in the milling process.

SRK (2000) measured the acid-generating capacity of 14 tailings samples and concluded that (1) wet sand tailings have no potential to generate acid, (2) some dry sand tailings can generate acid, and (3) most silt tailings do not have the potential to generate acid.

5.4.2.4 Remobilization

As noted above, the tailings pile solids represent a potential source of continued ground water contamination if contaminants are likely to be leached from them and enter the ground water over time. Several studies have focused on evaluating the potential for the tailings to leach over time.

SMI (2001; Appendix B−4) performed sequential extractions of tailings to evaluate relative leachability and chemistry of different mineral phases (Section 5.4.2.1). DOE (2003c) performed column tests on tailings from the base of the pile to evaluate the potential for contaminant release with the inundation of tailings with a rise in water table level. Those studies have demonstrated that leachable quantities of constituents remain in the tailings. As expected, constituents such as sulfate that are typically contained in more soluble minerals leach more readily. Metals are more recalcitrant. Effects of pile inundation on ground water quality are further evaluated in the ground water flow and transport section (Section 7.0)

Sediment samples collected from beneath the tailings pile are elevated in most site-related constituents compared to concentrations detected in background samples, indicating that some tailings pile contamination has migrated beneath the pile (see Section 4.8). However, with increasing depth, most concentrations of constituents decrease to background levels at depths of less than 10 ft below the base of the pile. For two of the three subpile boreholes included in the current study, radium-226 concentrations decreased to background within a 6-ft distance. ORNL data also confirm a sharp decrease in radium-226 concentrations below the tailings.

These results suggest that if leachable constituents are being removed from the tailings pile, most of the contamination is partitioning to the ground water system and not adsorbing to the subpile sediments. This means that only limited subpile soil cleanup would probably be required if the tailings pile were to be removed to another location. Only arsenic exceeded its soil screening level for protection of ground water. However, arsenic is detected only sporadically in ground water; the highest concentrations are in samples from the wood chip area located east of the tailings pile. This suggests that subpile sediments are not significantly affecting ground water quality with regard to arsenic. It appears that if soils were cleaned up to meet radium-226 cleanup standards, soils would be protective of ground water for other nonradiological constituents. Although the characterization data are considered sufficient for developing the conceptual site model, it may not be sufficient for estimating quantity of contaminated soil to be removed if the pile is relocated. Therefore, the extent of radium-226 contamination may need to be reassessed at a later date depending on project direction. If the decision is made to cap the pile in place, study results suggest that subpile sediments will have little impact on ground water quality. However, the high contaminant concentrations obtained from leaching the tailings may indicate that rewetting and draining of the pile itself has the potential to negatively impact ground water quality.

An issue of concern for future tailings management is whether uncovering the tailings will change the leaching characteristics. For example, uranium minerals that are chemically reduced will become soluble if exposed to oxidizing conditions. Exposure of the tailings to the atmosphere after removal of the cover could increase the oxidation state of the tailings.

Although heterogeneous, the tailings are probably dominantly oxidized, and it is unlikely that reduced uranium minerals (or other redox contaminants) are present. Therefore, uncovering the tailings should not significantly increase the concentrations of uranium (or other redox constituents) in the pore fluid.

Water that enters the tailings from precipitation or dust control is likely to become contaminated at approximately the same concentrations as currently exist in the pore fluids. Dissolution of soluble salts in the tailings will contribute sulfate, ammonia, and other ions. Uranium concentrations are likely to be controlled by oxidized uranium minerals or adsorption, which should equilibrate with added water. Thus, contribution to ground water contamination from tailings during excavation activities is likely to be more a function of the water flux through the system than it is to changes in chemical oxidation state. Ground water contamination during construction is best minimized through judicious water management.

5.4.3 Summary

Pore fluids in the tailings pile are consistently elevated in ammonia, molybdenum, sulfate, TDS, and uranium. Some samples are also high in arsenic, cadmium, copper, manganese, nitrate, selenium, and vanadium. The solid tailings material is consistently high in ammonia, arsenic, molybdenum, radium-226, sulfate, uranium, and vanadium. Some tailings samples also contain relatively high concentrations of cadmium, copper, and manganese. Leachability testing of samples from the tailings pile indicate that ammonia, sulfate, and uranium are present in leachable forms in the tailings pile and can be expected to persist in the pile until a large number of pore volumes of water have passed through them. Results of subpile sediment sampling indicate that the majority of contaminants that seep from the pile remain in solution and reach the ground water. A relatively small proportion is adsorbed to the soil beneath the pile.

5.5 Nature and Extent of Ground Water Contamination

Environmental ground water chemistry data associated with the Moab site is extensive and has been collected since the early 1970s as a requirement of Atlas Minerals operating license (see Section 3). Characterization of the ground water system as a component of site reclamation began in 1989 with the Corrective Action Plan (Western Technologies, Inc. 1989) and continued through the Trustee Studies in 2000 (SRK 2000 and SMI 2001) and the site investigations performed by DOE (see Section 4.0). SMI water quality results (SMI 2001) and the DOE water quality results obtained from the five quarterly sampling events conducted between December 2001 and December 2002 for all the ground water monitor wells that are completed in the basin-fill aquifer and located on site and immediately downgradient of the site boundary are summarized in Table 5–17. The range in natural background for each constituent, based on the background water quality results previously presented in Section 5.3 for the fresh Qal and brine Qal hydrochemical facies, is included in Table 5–17 for comparison. Details regarding the data collection and analyses of the ground water samples were previously reviewed in Section 4.0. Results of the individual analyses are presented in Appendix C. Sample locations are shown in Figure 5–27 and Plate 1.

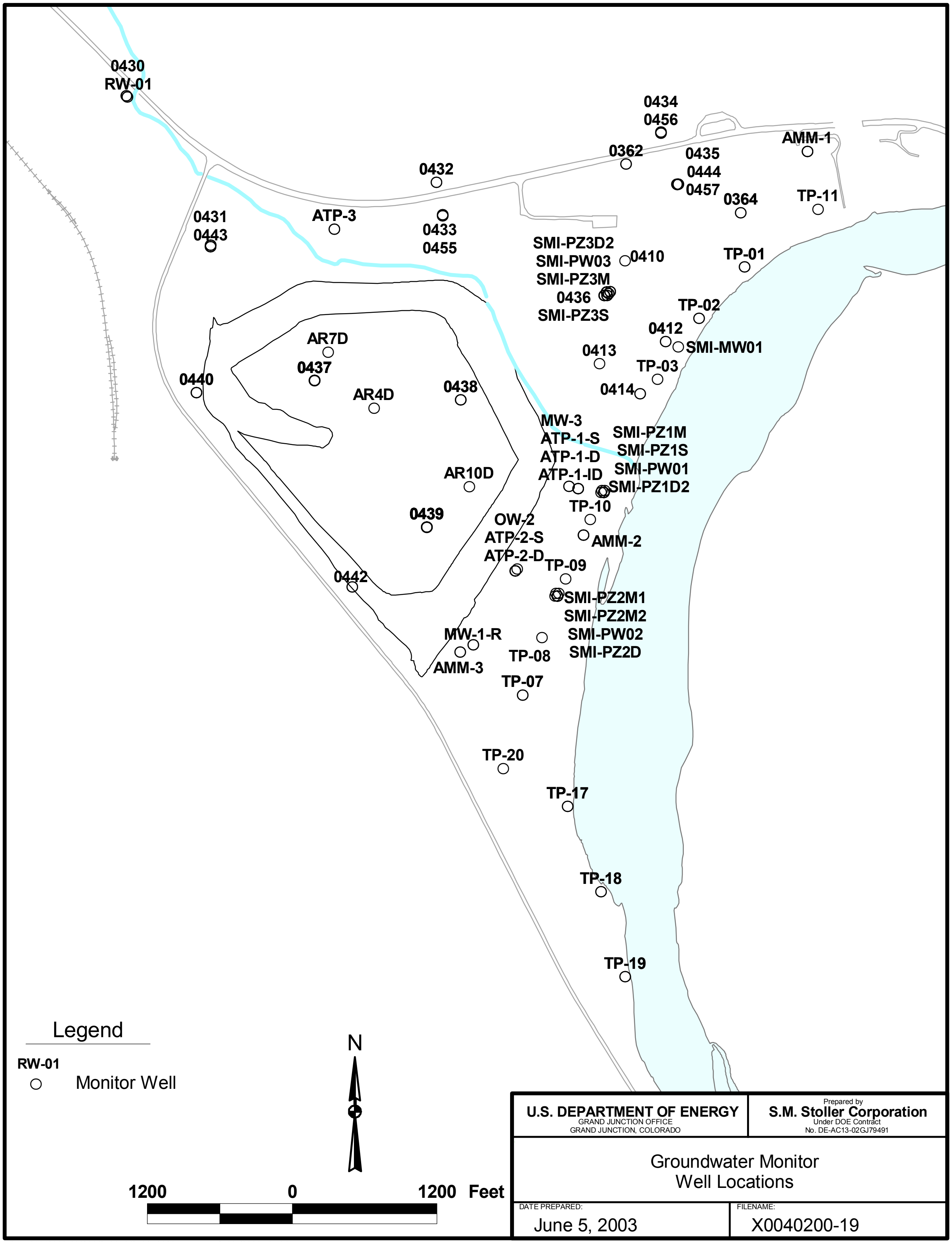

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Figure 5–27. Monitor Well Locations

Table 5–17. Comparison of On-Site and Downgradient Ground Water to Natural Background Ground Water Quality in the Basin-Fill Aquifer

Analyte	Unit	On-Site and Downgradient				Fresh Qal		Brine Qal	
		FOD	Mean	Range	Max Well	Background Range	Percent Exceeding Background	Background Range	Percent Exceeding Background
Major									
Alkalinity, total as $CaCO_3$	mg/L	109/109	549	35−1915	0442	137−189	77	109−733.04	46
Ammonia, total as N	mg/L	153/161	508	<0.003−4600	SMI-PZ2M2	<0.003−<1	86	0.0288−3	78
Calcium	mg/L	140/150	541	14.4−1470	0436	<47.4−294	74	1260−2370	12
Chloride	mg/L	232/232	13300	148−67800	ATP-1-S	135−4050	30	17154−52388	8
Magnesium	mg/L	147/152	511	24.6−1790	MW-1-R	<30.7−188	84	1000−2290	0
Nitrate as NO_3	mg/L	66/76	180	<0.02−2190	MW-1-R	1.22−15.9	61	<0.02−0.075	86
Potassium	mg/L	141/152	289	3−1400	ATP-2-D	9.29−56	70	59.4−150	51
Sodium	mg/L	160/168	9250	<293−36000	ATP-1-D	<128−2080	70	6370−18000	14
Sulfate	mg/L	233/234	6600	<173−23300	MW-1-R	180−1140	91	2004−6000	55
Metals									
Aluminum	mg/L	53/118	0.0207	0.002−0.29	SMI-PZ1D2	<0.0076−<0.051	11	<0.03−<0.03	24
Antimony	mg/L	15/69	0.000534	<0.0001−<0.0029	TP-01	<0.0001−<0.011	0	<0.0005−<0.05	0
Arsenic	mg/L	79/110	0.0109	<0.0001−0.361	0414	0.00018−0.0015	35	0.00015−0.11	3
Barium	mg/L	100/101	0.0362	<0.01−0.108	TP-18	0.0222−0.033	20	0.031−0.121	0
Beryllium	mg/L	1/6	0.000775	<0.001−0.0021	AR10D	0.002−0.002	17	0.0016−0.00165	17
Bismuth	mg/L	0/6	0.00158	<0.001−<0.011	TP-02	<0.011−<0.011	0		
Cadmium	mg/L	72/116	0.0018	<0.0001−0.0208	MW-1-R	<0.0001−<0.0017	31	<0.0001−0.014	2
Chromium	mg/L	2/69	0.000638	<0.0005−<0.003	ATP-1-S	<0.0005−<0.011	0	<0.003−<0.01	0
Cobalt	mg/L	61/75	0.00755	0.00055−0.064	AR10D	<0.0013−0.002	75	0.0451−0.0661	0
Copper	mg/L	65/110	0.0102	<0.0004−0.068	TP-18	<0.0004−0.005	42	<0.005−0.007	39
Iron	mg/L	42/96	1.28	<0.0008−17.1	AMM-3	<0.0008−<0.05	34	0.129−22.3	0
Lead	mg/L	6/76	0.000335	<0.0001−<0.0055	TP-02	<.0001−<.0055	0	0.00054−0.184	0
Lithium	mg/L	49/53	0.373	0.0201−1.71	MW-1-R	.0278−1	4	0.0579−0.335	36
Manganese	mg/L	157/169	3.1	<0.01−14.5	SMI-PZ1D2	<0.0001−0.0157	93	0.369−38.5	0
Mercury	mg/L	5/70	0.000488	<0.0001−0.003	TP-17	<0.0001−<0.0002	7	<0.0002−<0.0002	7
Molybdenum	mg/L	160/186	0.844	<0.001−10.8	AR7D	<0.0018−0.01	84	<0.004−<0.009	84
Nickel	mg/L	49/61	0.0185	<0.0006−0.089	AR10D	<0.0006−0.015	48	<0.004−0.0647	2
Selenium	mg/L	89/126	0.032	<0.0001−0.205	0414	0.0091−0.0266	23	<0.0001−0.009	40
Silver	mg/L	2/70	0.000309	<0.0001−<0.0055	TP-02	<0.0001−<0.0055	0	<0.0001−<0.004	0
Strontium	mg/L	120/123	15	<1.08−38.17	SMI-PZ2D	<2.25−8.45	72	23.2−65	0
Thallium	mg/L	6/68	0.000451	<0.0001−<0.011	TP-02	<0.0001−<0.011	0	<0.00014−<0.00014	9
Uranium	mg/L	226/231	2.76	<0.0001−23.3	TP-02	0.0042−0.0259	86	0.0007−0.0269	86
Vanadium	mg/L	64/137	0.154	<0.0003−7.1	0414	0.00061−0.0164	15	<0.0015−0.135	9
Zinc	mg/L	71/100	0.0129	<0.0006−.16	AR10D	<0.0006−0.011	25	0.0179−0.0261	10
Other									
Boron	mg/L	82/83	0.999	<0.1−4.21	0436	0.106−0.477	58	0.181−1.33	11
Dissolved Oxygen (unfiltered)	mg/L	72/72	1.34	0.13−10.38	0412	0.9−6.94	1	0.13−1.84	31
Fluoride	mg/L	39/49	1.79	<.151−10.3	ATP-1-D	<0.193−1.46	29	1.85−3.37	12
ORP (unfiltered)	mV	141/141	-9.67	-443−278	SMI-PZ3D2	66.9−619		-184−206	
Silica	mg/L	13/13	18.6	11.5−27.6	AR4D	14.3−14.3	77		
Total Dissolved Solids	mg/L	175/175	32700	1440−111000	ATP-1-D	677−7820	70	31214−97014	7
pH (unfiltered)	s.u.	269/269	7.2	6.5−8.4	SMI-PZ3S	7.19−8.11	1	5.7−7.18	43
Physical									
Density	g/cm^3	37/37	1.02	0.9982−1.0618	SMI-PZ2D	1.0002−1.0002	95	1.0195−1.0645	0
Specific Conductance (unfiltered)	µmhos/cm	222/222	34700	18.51−133476	ATP-1-D	1078−14040	68	43695−111287	10
Specific Gravity		35/35	1.03	1−1.08	TP-19	1−1.01	89	1.04−1.04	29
Temperature (unfiltered)	°C	192/192	16.9	12.42−25	SMI-PZ2D	14.3−19.4	14	8.73−16.8	52
Turbidity (unfiltered)	NTU	96/96	51.4	0−1000	0440	.2−144	5	.82−373	3
Radionuclides									
Gross Alpha	pCi/L	68/85	1970	<15.02−19016.16	TP-02	<6.73−<73.92	80	<356.33−<473.08	74
Gross Beta	pCi/L	62/85	708	<16.01−5292.71	TP-02	7.39−64.4	71	<299.19−<444.99	48
Radium-226	pCi/L	45/88	0.684	<0.08−7.9	AR4D	.07−.16	42	<.29−9.26	0
Radium-228	pCi/L	36/82	2.55	<0.64−15.91	ATP-1-ID	<.5−1	41	2.6−6.09	9
Thorium-230	pCi/L	10/70	0.861	<0.2−<5.8	MW-3	<1.5−<1.6	6	<1.6−<1.6	6

< less than detection limit; one-half the detection limit used to calculate the mean.
FOD - frequency of detection

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Ammonia (total as N), nitrate (as NO_3), sulfate, molybdenum, uranium, and gross alpha are the site-related constituents most prevalent in the basin-fill aquifer, as indicated by the relatively high frequency (greater than 50 percent) of samples with concentrations that exceed the upper limit of the range in natural background for both the fresh Qal and brine Qal facies. The relatively low site-specific R_d measured for uranium and ammonia (Section 4.7) explains the more prevalent occurrence of these site-related constituents, which are conserved in the ground water and more easily dispersed from the source area. Similarly, studies at other UMTRA sites indicate that molybdenum and nitrate are geochemically conservative and tend to be highly mobile in ground water under almost all conditions.

Concentrations of some constituents, such as magnesium, cobalt, manganese, and strontium, exceed the upper limit of the range in natural background for the fresh Qal facies by more than 50 percent but do not exceed the upper limit of the natural background range for the brine Qal facies in any of the samples. Other site-related constituents are also present at concentrations above the upper limit of the natural background range for the fresh Qal facies, but only exceed the upper limit of the background range for the brine Qal facies in a few samples. For example, cadmium and nickel concentrations exceed the upper limit in natural background for the fresh Qal facies in 31 and 48 percent of the samples respectively, but only exceed the upper limit in natural background for the brine Qal facies in 2 percent of the samples. Arsenic levels exceed the upper limit of the range in approximately 35 percent of the samples when compared to the fresh Qal background but exceed background in only 3 percent of the samples when compared to the brine Qal background. The low frequency that these constituents exceed the upper limit of the background range for the brine Qal facies reflects the relatively high concentrations that are naturally occurring in the Paradox Formation brine, as discussed in Section 5.3.

Concentrations of other constituents, such as selenium, exceed the background range in approximately 23 percent of the samples when compared to the fresh Qal facies background and in 40 percent of the samples when compared to the brine Qal background. Copper concentrations exceed the background range in 42 and 39 percent of the samples when compared to the fresh Qal and brine Qal facies, respectively. Antimony, chromium, lead, and silver concentrations do not exceed the natural background range for either the fresh or brine Qal facies in any of the samples.

Ground water concentrations of arsenic, barium, cadmium, chromium, lead, mercury, molybdenum, nitrate, selenium, silver, uranium (combined U-234 and U-238), gross alpha (excluding radon and uranium), and radium (combined radium-226 and radium-228) are regulated by EPA standards in 40 CFR 192 (Table 2–1). Of these constituents, the maximum concentrations detected for arsenic, cadmium, uranium, radium, gross alpha, nitrate, selenium, and molybdenum exceed their respective standards. The remaining regulated constituents—barium, chromium, lead, mercury, and silver—are all present at relatively low concentrations below their respective EPA ground water standard.

Ammonia, arsenic, cadmium, uranium, radium-226 and -228, gross alpha, nitrate, selenium, and molybdenum are selected for further discussion in the following sections. Arsenic, cadmium, uranium, radium, gross alpha, nitrate, selenium, and molybdenum are selected because they are site-related constituents with concentrations that exceed their respective EPA ground water standards. Although no EPA drinking water or ground water standard exists for ammonia, it is selected for further analysis because it is present at concentrations significantly greater than natural background, it is one of the most prevalent contaminants in site ground water, and it is the constituent of greatest ecological concern in ground water discharging to the Colorado River in backwater areas adjacent to the site.

5.5.1 Distribution of Inorganic Contamination in the Ground Water

The following discussion is based on analytical results of the most recent ground water sampling. In most cases, the most recent result is from the December 2002 sampling, which provides the most extensive and comprehensive coverage from the permanent monitor wells to date. Grab samples collected from temporary wells during the drilling campaign in July and August 2002 are also included to profile the chemical distributions as a function of depth in the aquifer. In a few cases, earlier data collected in 2000 by SMI (Section 3) are used to provide coverage in areas of the site where monitor wells were abandoned before DOE could sample the locations. These sample results are mostly related to the temporary wells installed by SRK (2000) in the basin-fill aquifer beneath the tailings pile and the temporary wells installed by ORNL (1998a) along the bank of the Colorado River. Sample locations are shown in Figure 5–27 and Plate 1.

EVS three-dimensional modeling software (http://www.ctech.com) was used to krige and contour the most recent ground water chemical results for ammonia, uranium, gross alpha, nitrate, selenium, and molybdenum. The EVS three-dimensional model allows visualization of the contoured surface at discrete elevations in the aquifer. Contours of contaminant concentrations interpolated on the upper surface of the alluvial ground water (first water) and at selected depths below the water surface are presented. Using the most recent ground water chemistry results provides the most current snapshot of the extent of contaminant distributions in the aquifer. Since only the most recent results are presented in the following figures and tables, the maximum value at a location may vary from the maximum value previously presented in Table 5–17, which is a summary based on all water quality results obtained between 2000 and 2002.

5.5.1.1 Ammonia

The areal distribution of ammonia concentrations greater than 50 mg/L, interpolated and contoured on the upper surface of the ground water, is presented in Figure 5–28. The highest concentrations in the shallow ground water, greater than 500 mg/L, appear near the downgradient edge of the toe of the tailings pile and extend to the Colorado River. Some of the ammonia concentrations in the shallow ground water in this area of the site are most likely associated with seepage from the former toe drains. These drains are identified in Figure 3–4 as drain sumps and historically were referred to as the north sump and the south sump.

A similar distribution of ammonia concentrations near the toe of the tailings pile is evident at 50 ft below the surface of the ground water as shown in Figure 5–29. The maximum concentration of 4,600 mg/L was detected in a grab sample at monitor well SMI-PZ2M2 (February 2002) at approximately 45 ft below the surface of the ground water. Monitor well SMI-PZ2M2 is located between the toe of the tailings pile and the Colorado River. At the 50-ft depth, an ammonia plume having concentrations greater than 50 mg/L is also present in the former millsite area.

At 150 ft below the surface of the ground water the elevated ammonia concentrations are no longer present in the former millsite area (Figure 5–30). Ammonia concentrations also decrease at the toe of the pile. Conversely, concentrations increase beneath the center of the tailings pile, indicating the presence of a deeper ammonia plume. This deeper plume extends approximately 200 ft below the surface of the ground water, as shown in the cross section in Figure 5–31.

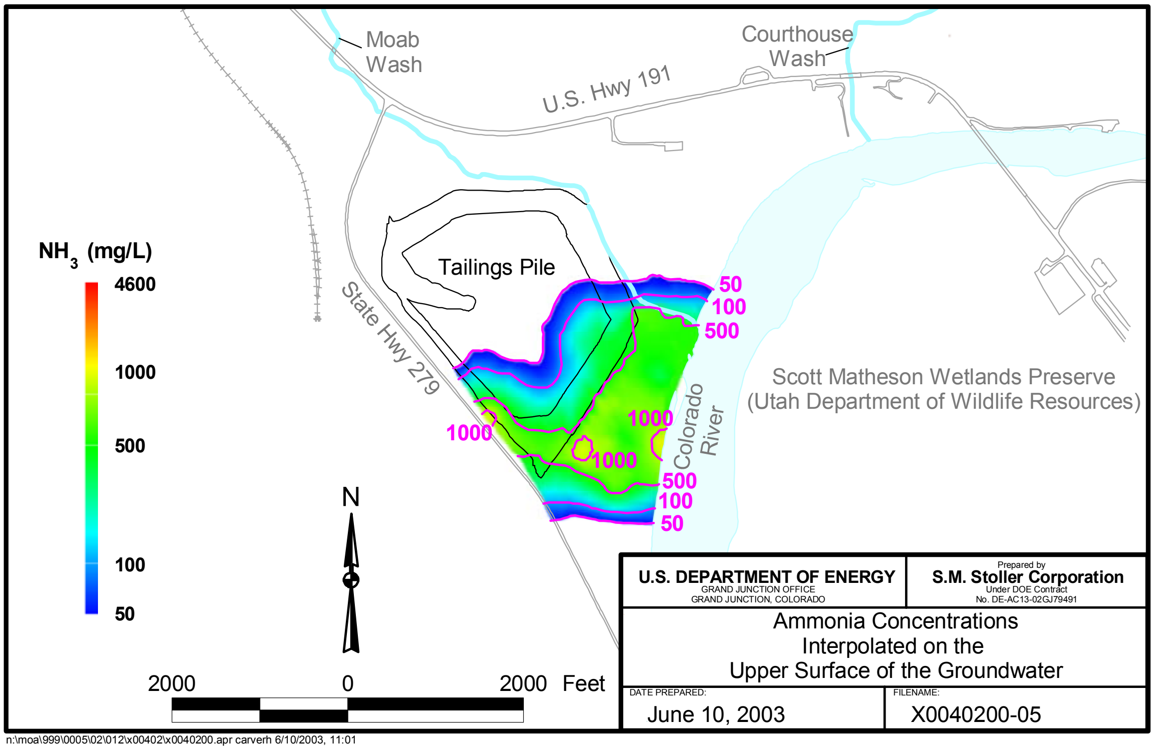

Figure 5–28. Ammonia (total as N) Concentrations Interpolated on the Upper Surface of the Ground Water

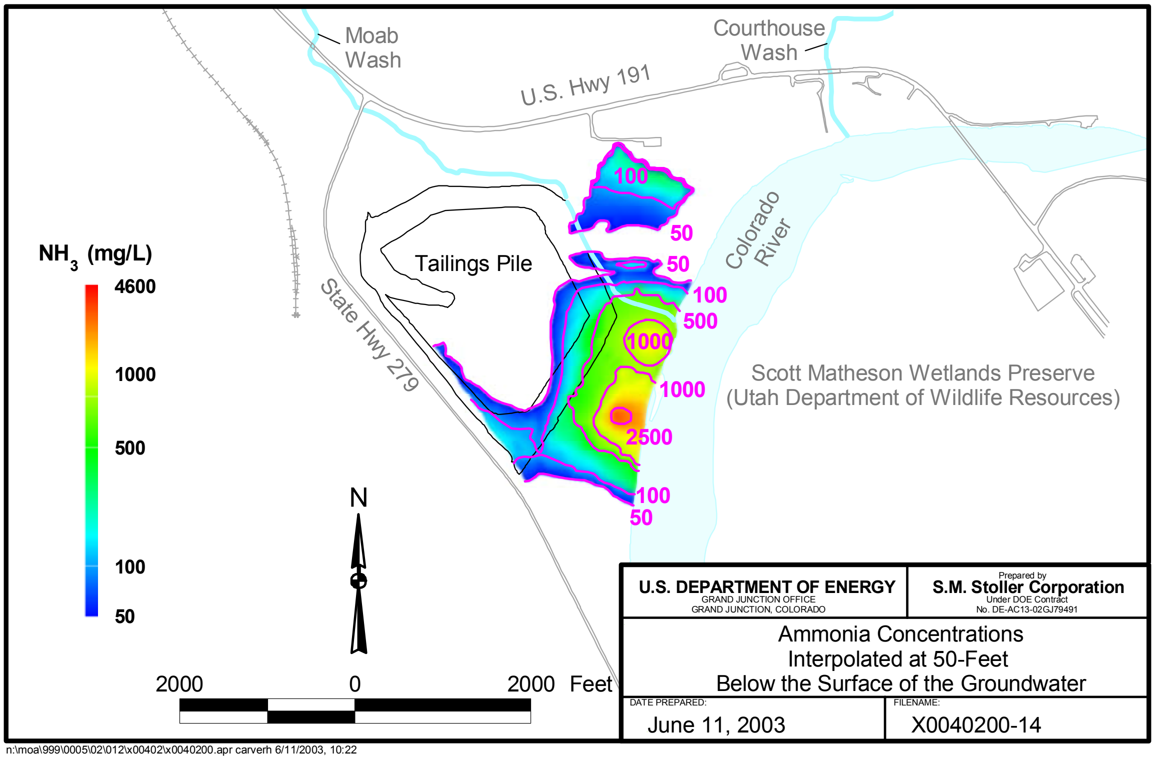

Figure 5–29. Ammonia (total as N) Concentrations Interpolated at 50 ft Below the Surface of the Ground Water

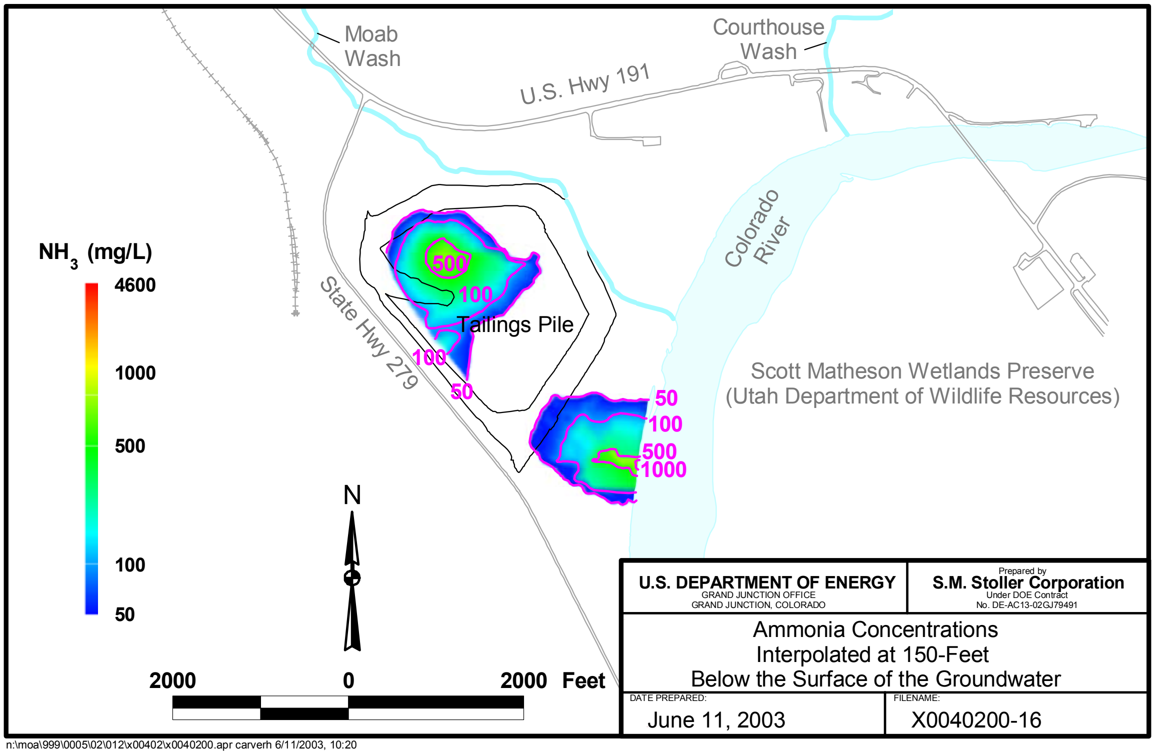

Figure 5–30. Ammonia (total as N) Concentrations Interpolated at 150 ft Below the Surface of the Ground Water

Ammonia concentrations in the shallow ground water beneath the center of the tailings pile, as evidenced by the results posted for monitor well MOA-437 in Figure 5–31, are relatively low (less than 5 mg/L). This suggests the deep plume is mature and probably associated with former seepage of high-density fluids from the tailings pond during the earlier milling operations (Section 3-8) that has infiltrated the deeper brine Qal facies. The highest ammonia concentration of 1,630 mg/L detected in monitor well MOA-437 is located within the higher density brine Qal facies that underlies the site. TDS at this sample depth is 66,400 mg/L, as evidenced in the cross-section presented in Figure 5–21.

During milling, the tailings pile may have contained fluids with TDS ranging from 50,000 to 150,000 mg/L. Because these salinities exceed the 35,000 mg/L concentrations at the saltwater interface, they are believed to have had sufficient density to migrate vertically downward into the brine. This vertical migration of the tailings pore fluids into the saltwater system is believed to have created a reservoir of ammonia that now resides below the saltwater interface. This ammonia plume below the interface probably came to rest at an elevation where it was buoyed by brine having a similar density. Under present conditions, the ammonia plume beneath the saltwater interface represents a long-term source of ammonia to the upper alluvial ground water system. The ammonia source at the saltwater interface (basal or ammonia flux), the legacy plume, and seepage of ammonia concentrations from the tailings pore fluids are illustrated in the conceptual model presented in Figure 5–32.

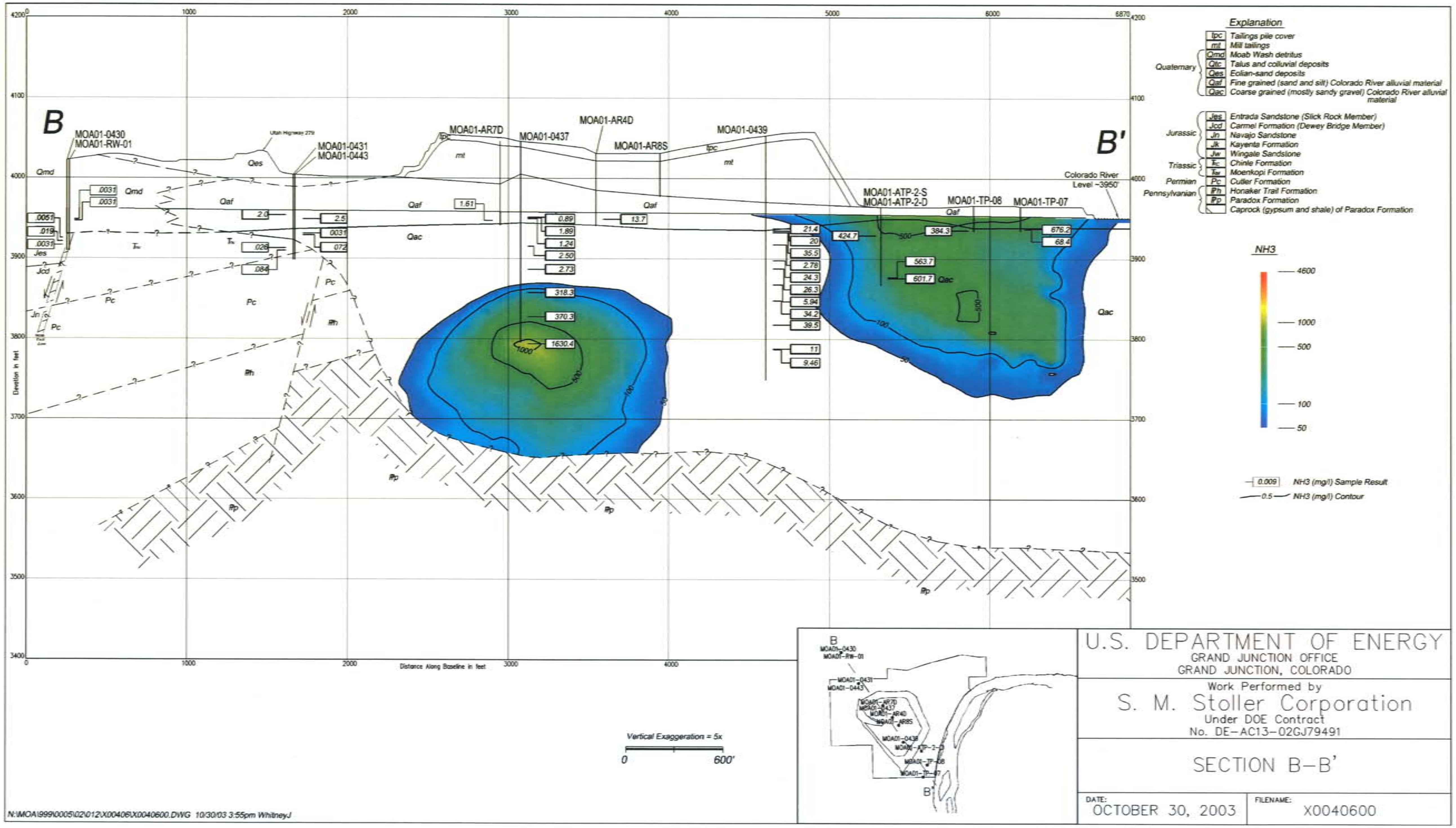

Figure 5–31. Cross-Sectional View B–B' of Ammonia (total as N) Concentrations in Ground Water Beneath the Tailings Pile Area

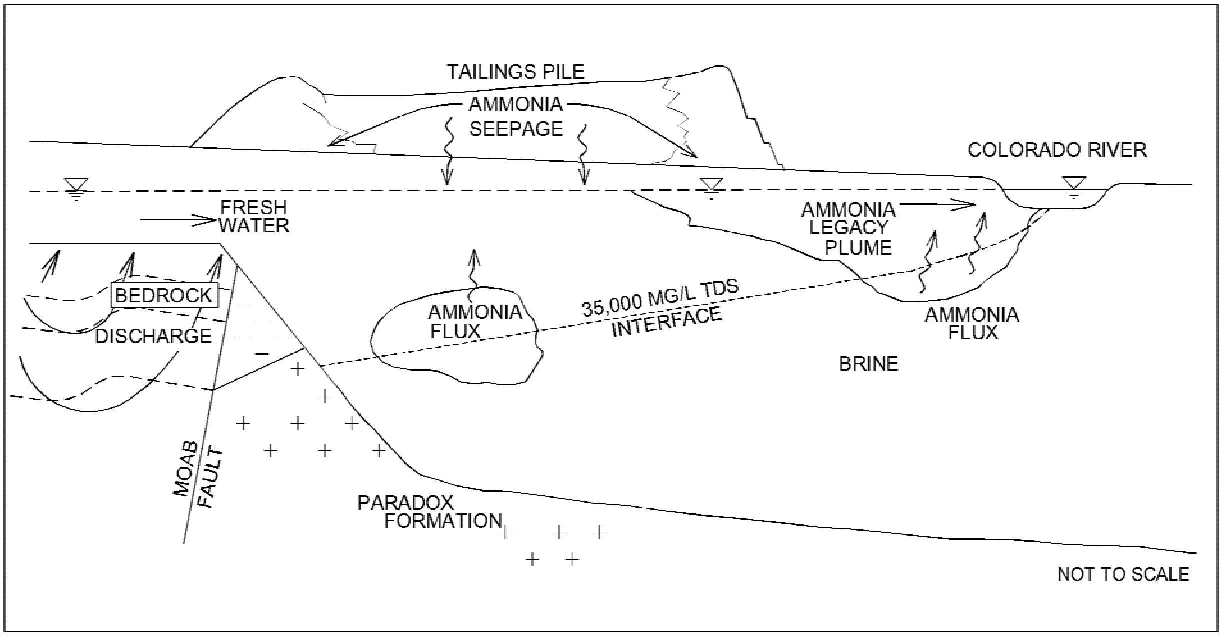

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Figure 5–32. Conceptual Model of Seepage of Ammonia Concentrations from the Tailings Pore Fluids

Since the release of tailings pond fluids containing high TDS concentrations infiltrated the ground water during milling operations, the volume of relatively fresh water entering the site upgradient of the tailings pile may have diluted the ammonia levels in the shallow ground water. Advective flow of fresh water through the higher-density fluids is insignificant, and thus the ammonia concentrations persist at depth. Oxidation of ammonia to nitrate or nitrogen may also contribute to lower ammonia concentrations observed in the upgradient shallow ground water beneath the tailings pile where aerobic conditions are more likely.

Ammonia Plume Volume

EVS three-dimensional modeling software was also used to estimate the volume of the ammonia plume contained in the basin-fill aquifer for each of the three hydrochemical facies. Estimated volumes for each hydrochemical facies are presented in Table 5–18. Most of the aquifer (82 percent) is characterized by ground water containing TDS concentrations greater than 10,000 mg/L (brackish and brine facies). The volume of the ammonia plume contained below the saltwater interface (>35,000 mg/L TDS brine facies) is estimated at 1.33 billion gallons. This volume represents approximately 60 percent of the total ammonia plume (2.26 billion gallons).

Table 5–18. Estimated Volume of Ammonia Plume in the Hydrochemical Facies at the Moab Site

TDS (mg/L) Facies	Volume of Basin-Fill Material (gallons)	Porosity	Volume of Ground Water (gls) in Aquifer	Percent of TDS Facies in Aquifer	Volume (gallons) of Ammonia Plume >50 mg/L
<10,000	5,196,478,800	0.30	1,558,943,600	18	60,149,700
>10,000	24,374,913,500	0.30	7,312,474,000	82	2,201,557,700
>35,000	17,527,645,400	0.30	5,258,293,600	60	1,331,118,000

Ammonia Concentration Trends

Ammonia concentrations in shallow ground water samples collected along the bank of the Colorado River and downgradient from the toe of the tailings pile were measured by ORNL in 1997 (ORNL 1998a). Additional ammonia sampling and analyses were conducted at some of the original ORNL locations and at several new locations established by SMI in 2000 (SMI 2001) and by DOE in 2001 (Section 4). Most of these water samples were collected from a screened interval approximately 15 to 20 ft below ground surface. Analytical results for the three sampling events are profiled in Figure 5–33. The most apparent pattern observed in the profile is a peak in concentration that occurs approximately 1,000 ft south of Moab Wash. Peak concentrations of 1,285 mg/L measured in 1997, 1,130 mg/L measured in 2000, and 1,360 mg/L measured in 2001 were detected in this area. Although the actual location of the peak seems to vary with the sampling event, the general shape and concentration of the peak appears relatively consistent over the 4 years of monitoring.

Some of the earliest ammonia concentrations measured in ground water samples at the site were collected in 1996 from monitor well ATP-2-S, which is located near the center of the toe of the tailings pile and approximately 700 ft upgradient of the peak concentrations shown in Figure 5–33. Monitor well ATP-2-S also has some of the greatest amount of historical ammonia data. These results are presented in the time-concentration plot in Figure 5–34 and suggests that ammonia concentrations are declining with time.

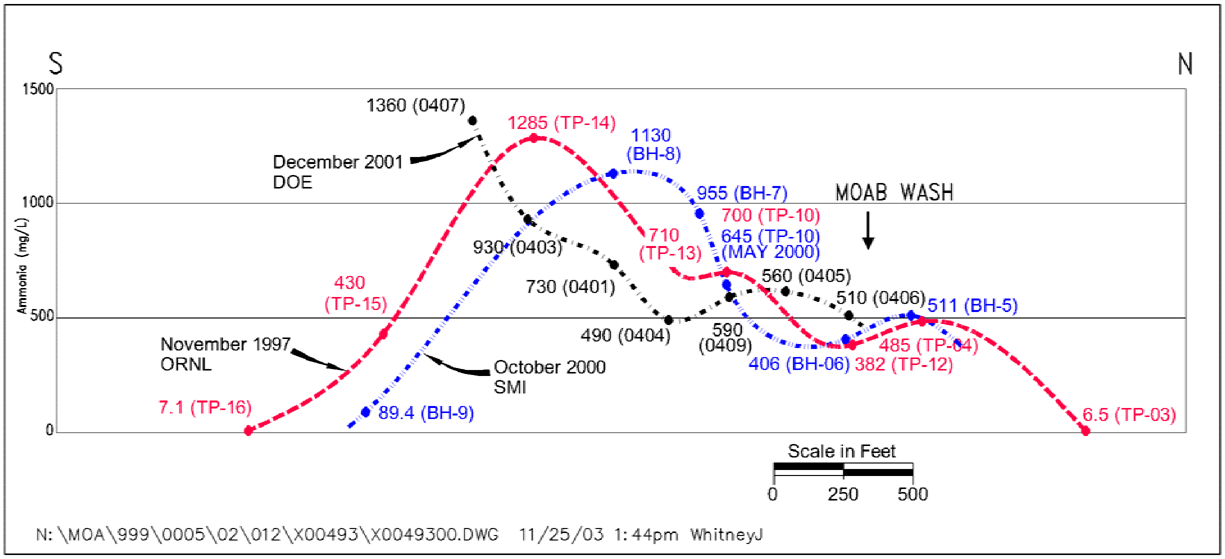


Figure 5–33. Ammonia (total as N) Concentration Profile in the Shallow Alluvial Aquifer Adjacent to the Bank of the Colorado River and Downgradient from the Toe of the Tailings Pile (Well locations designated in parentheses following sample result)

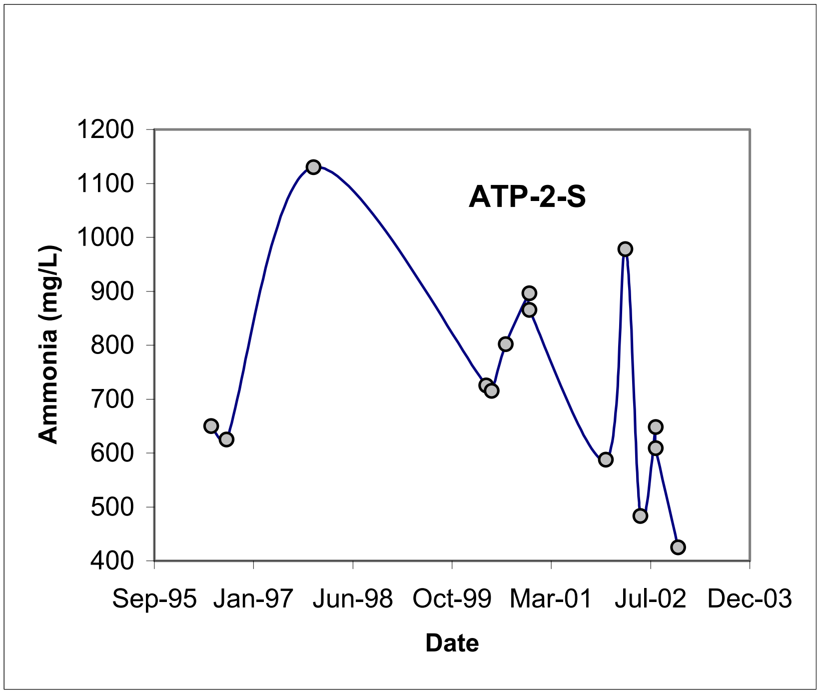


Figure 5–34. Ammonia Time-Concentration Plot for Monitor Well ATP-2-S Located at the Downgradient Toe of the Tailings Pile

5.5.1.2 Arsenic

Arsenic concentration exceeds the 0.05 mg/L EPA ground water standard at only one location in the former millsite area. Two samples collected from this location, MOA-414, from the shallow basin-fill aquifer contained 0.3 mg/L. The most recent results are summarized in Table 5–19. Monitor well MOA-414 is located in the area believed to be where the former wood chips were disposed of (shown in the southeast corner of Figure 3–5).

Table 5–19. Summary of Arsenic Results that Exceed the 0.05 mg/L Standard

Location	Sample Depth Below Ground Surface (ft)	mg/L
414	6.53	0.313
414	9.8	0.315

5.5.1.3 Cadmium

Cadmium concentrations exceed the 40 CFR 192 standard at only two locations: MOA-437 and MW-1-R. Both locations are in the tailings pile area. MOA-437 is completed in the shallow basin-fill aquifer beneath the pile, and MW-1-R is completed in the shallow basin-fill aquifer at the toe of the pile. The maximum cadmium concentration of 0.02 mg/L was detected at both MOA-437 and MW-1-R; this value only slightly exceeds the 0.01 mg/L standard. Results are summarized in Table 5–20.

Table 5–20. Summary of Cadmium Results that Exceed the 0.01 mg/L Standard

Location	Sample Depth Below Ground Surface (ft)	mg/L
437	95	0.0113
437	95.65	0.0155
437	94	0.0202
MW-1-R	8	0.0208

5.5.1.4 Gross Alpha

Gross alpha concentrations exceeding the 15 pCi/L ground water standard in 40 CFR 192 are detected in numerous water samples. However, a large number of these results were reported below a detection limit that is much greater than the 15 pCi/L standard. The relatively high detection limit is due to interferences with high TDS concentrations in the sample from the brine Qal facies. The distribution pattern of gross alpha concentrations for those results that are above detection limit appear consistent with the pattern presented for uranium concentrations in Figure 5–39, and therefore a separate figure showing gross alpha is not presented.

5.5.1.5 Molybdenum

Molybdenum concentrations interpolated and contoured on the surface of the ground water are distributed in the basin-fill aquifer as shown in Figure 5–35. The most notable feature in Figure 5–35 is the area of relatively low concentrations in the center of the plume that is less than the 0.1 mg/L 40 CFR 192 standard. This area of low concentrations is observed in samples collected from surface of the ground water at wells AR10D and MOA-439. Molybdenum concentrations in these wells are reported at 0.03 mg/L and 0.06 mg/L, respectively. Relatively low concentrations in this area are also detected in grab samples collected at depth during installation of monitor well MOA-439, as evidenced from the concentrations that are contoured and plotted in the cross-section presented in Figure 5–36. The location and shape of the 0.1 mg/L molybdenum contour shown in the cross section at the base of the plume approximates the 35,000 mg/L TDS boundary shown in the TDS cross section presented in Figure 5–24.

The maximum molybdenum concentrations of 10.8 mg/L, 8.7 mg/L, and 8.5 mg/L are detected at locations AR7D, MW-1-R, and MOA-437, respectively. AR7D and MOA-437 are located at the upgradient end of the tailings pile. Concentrations in this area of the plume decrease rapidly with depth, as observed in the cross section (Figure 5–36). MW-1-R is located between the toe of the tailings pile and the Colorado River. Elevated concentrations at this location are probably related to seepage from the former south drain sump (Figure 3–4). Elevated concentrations of ammonia, nitrate, uranium, and selenium are also detected in the shallow ground water at this location. Finer overbank material deposited in this area of the site may explain the relatively higher concentrations that seem to persist in the immediate area surrounding monitor well MW-01-R. Lithologic information is not available for MW-1-R; however, examination of the borehole log prepared for nearby monitor well AMM-3 (Appendix A) indicates clay is present in the upper saturated formation. Lower permeability associated with clay would increase the time required to disperse the plume.

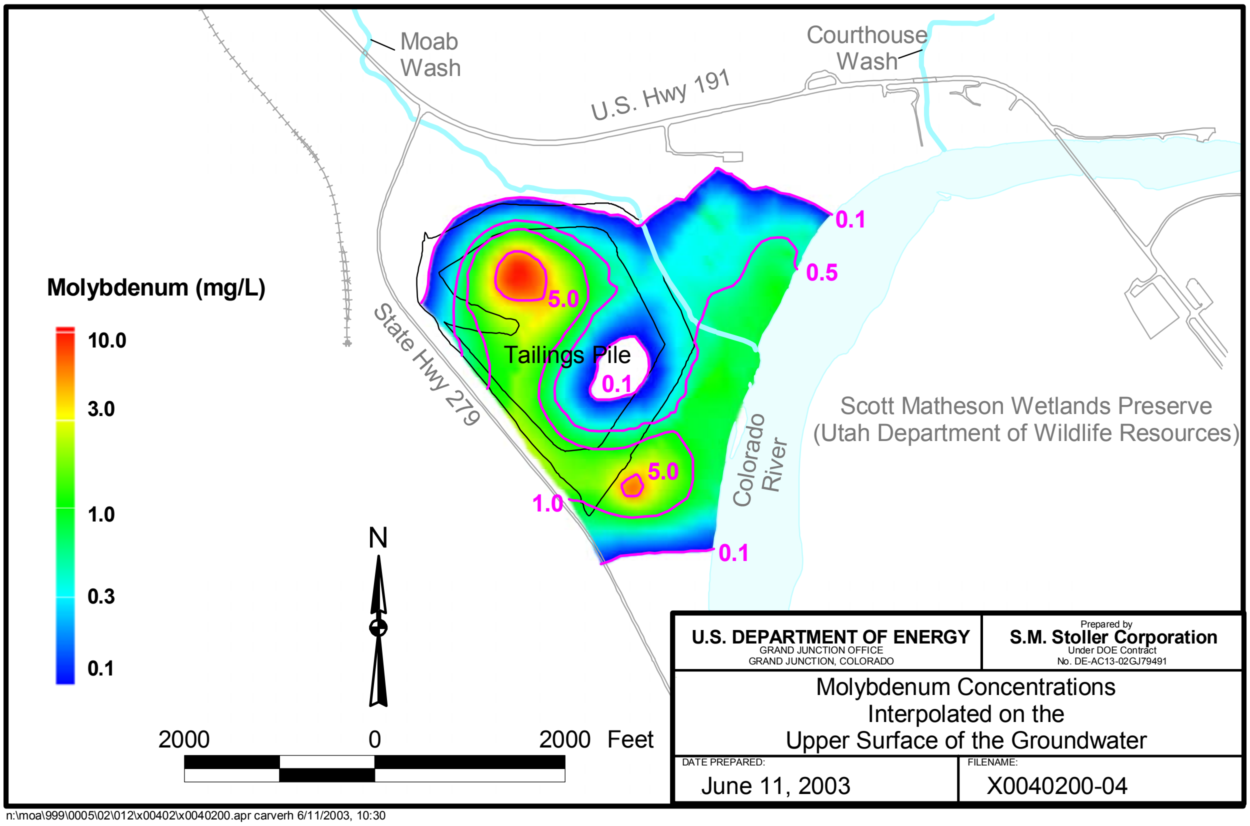


Figure 5–35. Molybdenum Concentrations Interpolated on the Upper Surface of the Ground Water

5.5.1.6 Nitrate

Nitrate concentrations in the ground water that exceed the 44 mg/L (nitrate as NO_3) EPA standard in 40 CFR 192 are shown in Figure 5–37. This distribution of nitrate concentrations interpolated on the upper surface of the ground water is similar to the distribution observed for ammonia (Figure 5–28). The highest nitrate concentrations are associated with monitor wells MOA-442 and MW-1-R, both located in the tailing pile area and completed in the shallow aquifer. MOA-442 is located along the southwest boundary of the pile, and MW-R-1 is located near the toe of the pile. Nitrate concentrations in these well were reported at 1,800 mg/L and 2,190 mg/L, respectively. All other nitrate concentrations in the tailings pile are less than 275 mg/L.

Nitrate concentrations are also elevated above the 44 mg/L standard in the former millsite area at three monitor well locations. These wells, MOA-412, -413, and –414, are located in the former wood chip disposal area and are screened at the surface of the ground water. Concentrations are reported at 192, 334, and 346 mg/L, respectively.

5.5.1.7 Radium-226 and -228

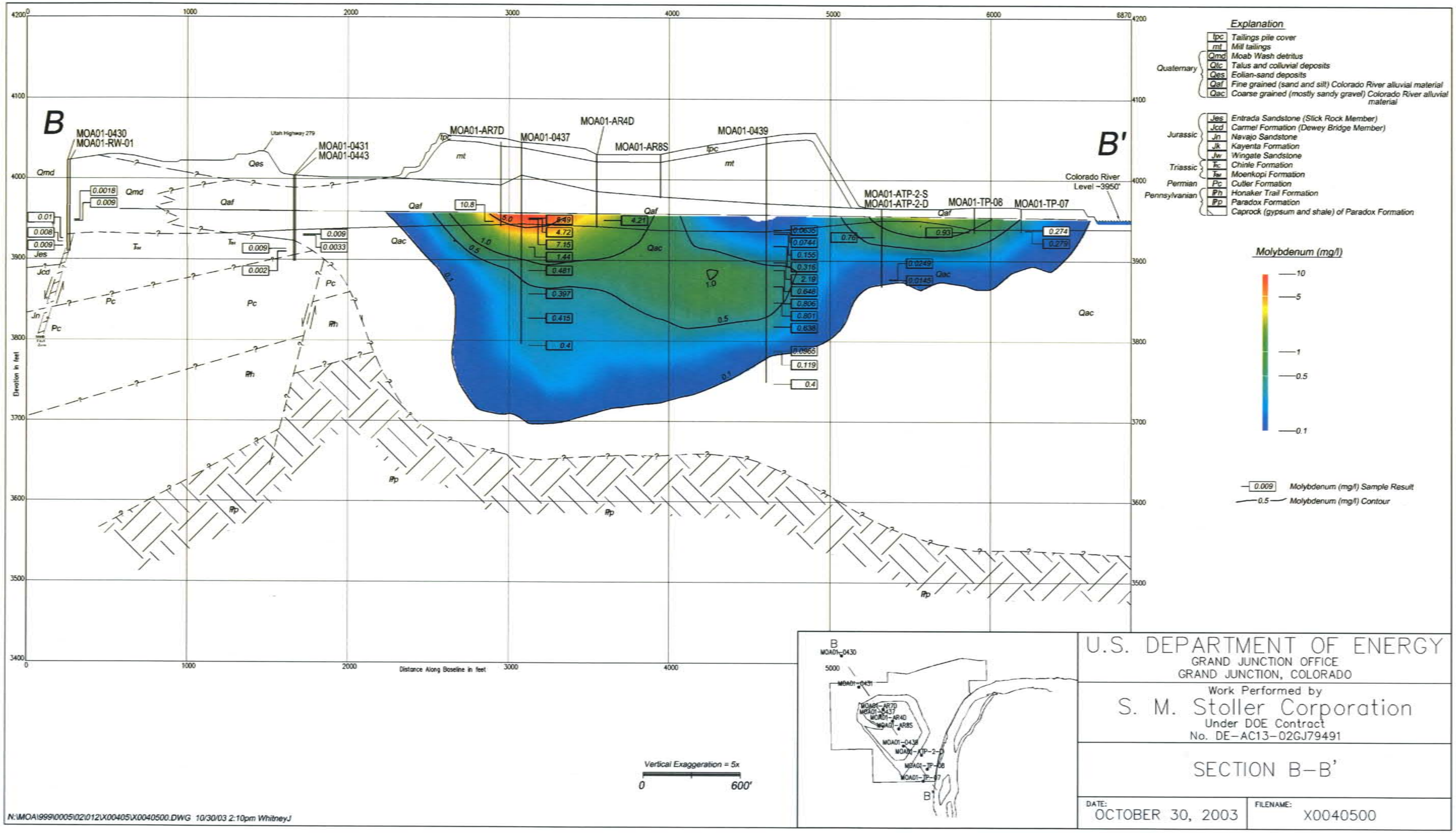
Radium concentrations (combined radium-226 and -228) exceed the 5 pCi/L 40 CFR 192 standard at eight locations: MOA-434, -435, -436, -437, -439, -444, ATP-1, and ATP-2. Results are summarized in Table 5–21. All elevated radium concentrations are believed to be the result of naturally occurring concentrations in the Chinle or Paradox Formations. For example, the slightly elevated radium concentrations of 8.9 pCi/L and 11.5 pCi/L were detected in water samples collected from on-site monitor well MOA-435, which is screened across the Chinle Formation. A background radium concentration of 5.2 pCi/L in the water sample collected from upgradient monitor well MOA-434, which is also screened across the Chinle Formation, also exceeds the 5 pCi/L standard. The remaining results presented in Table 5–21 that exceed the standard for radium are all associated with samples collected from monitor wells that are completed in the deeper brine Qal facies. These slightly elevated radium results, ranging from 5.2 pCi/L to 17.4 pCi/L, are believed to be natural contribution of radium-226 and -228 from the Paradox Formation. Most of the natural contribution appears to be from the radium-228 isotope.

Table 5–21. Summary of Radium Results that Exceed the 5 pCi/L Standard

Location	Sample Depth Below Ground Surface (ft)	pCi/L		
		Ra-226	Ra-228	Combined Ra-226+228
434	80.39	3.11	2.07	5.18
435	173.43	6.81	2.06	8.87
435	176	8.42	3.04	11.46
436	197.7	0.57	11.09	11.66
436	200	0.65	11.05	11.7
437	94	1.57	4.32	5.89
437	130	1.53	13.18	14.71
437	188	0.69	5.95	6.64
437	218	1.12	5.62	6.74
439	135	1.22	5.68	6.9
439	165	3.5	5.57	9.07
439	185	0.99	4.24	5.23
439	205	1.98	13.42	15.4
439	265	3.02	7.37	10.39
439	266	1.07	6.44	7.51
444	115	1.56	6.87	8.43
444	115.91	1.8	5.54	7.34
ATP-1-D	396.47	0.43	9.96	10.39
ATP-1-ID	307.33	1.5	15.91	17.41
ATP-1-S	150	1.07	10.95	12.02
ATP-2-D	87.5	0.3	10.39	10.69

5.5.1.8 Selenium

Concentrations of selenium in ground water exceed the 0.05 mg/L EPA Safe Drinking Water Act standard at 10 locations and are distributed in two separate plumes, which is shown on the contour map presented in Figure 5–38. One plume is located east of Moab Wash in the former millsite area, and the other is located on the west side of Moab Wash and is associated with the tailings pile. Elevated selenium concentrations associated with the east plume are limited to three monitor wells (MOA-412, -413, and –414) that are all screened across the uppermost surface of the basin-fill aquifer. These wells are located in the former wood chip disposal area. Selenium concentrations in ground water in this area of the site range from 0.09 mg/L to 0.17 mg/L, values that are only slightly above the 0.05 mg/L Safe Drinking Water Act standard.



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GRAND JUNCTION OFFICE
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Under DOE Contract
No. DE-AC13-02GJ79491

SECTION B–B'

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Figure 5–36. Cross-Sectional View B–B′ of Molybdenum Concentrations in Ground Water Beneath the Tailings Pile Area

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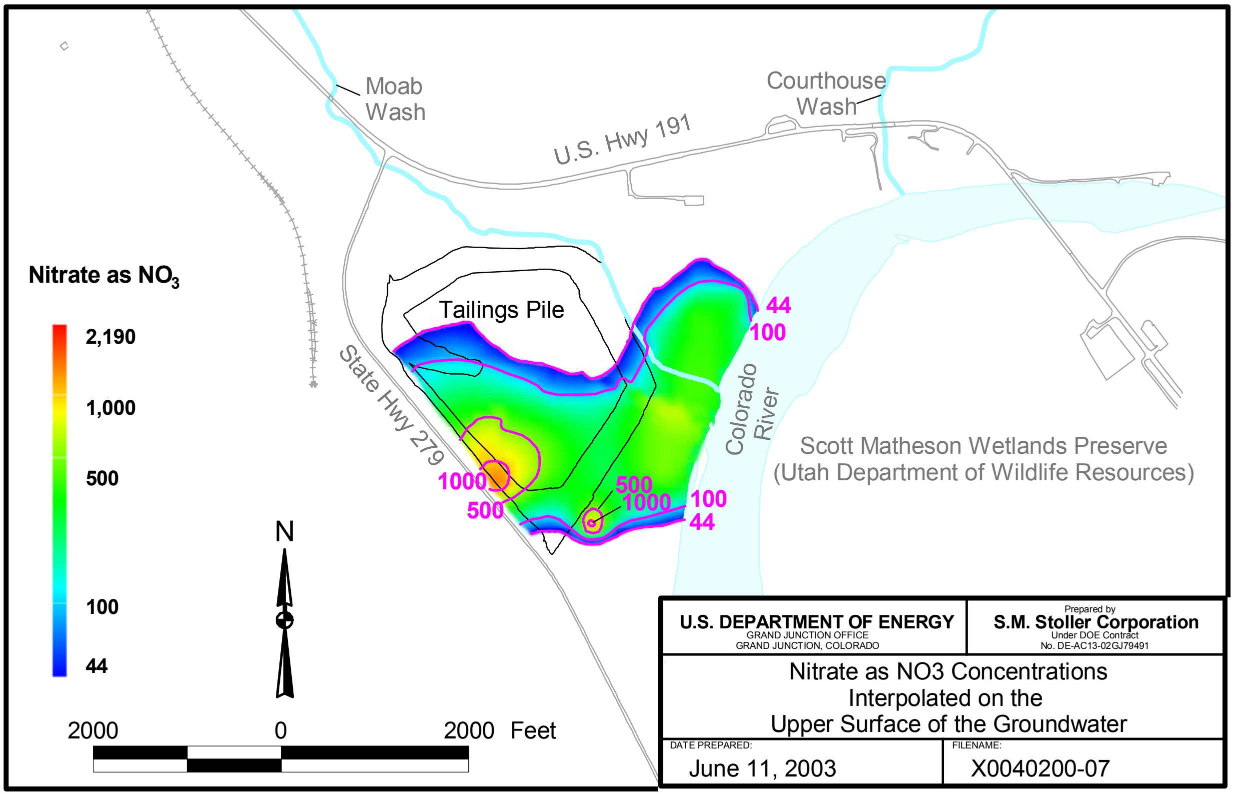

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Figure 5–37. Nitrate Concentrations Interpolated on the Ground Water Surface

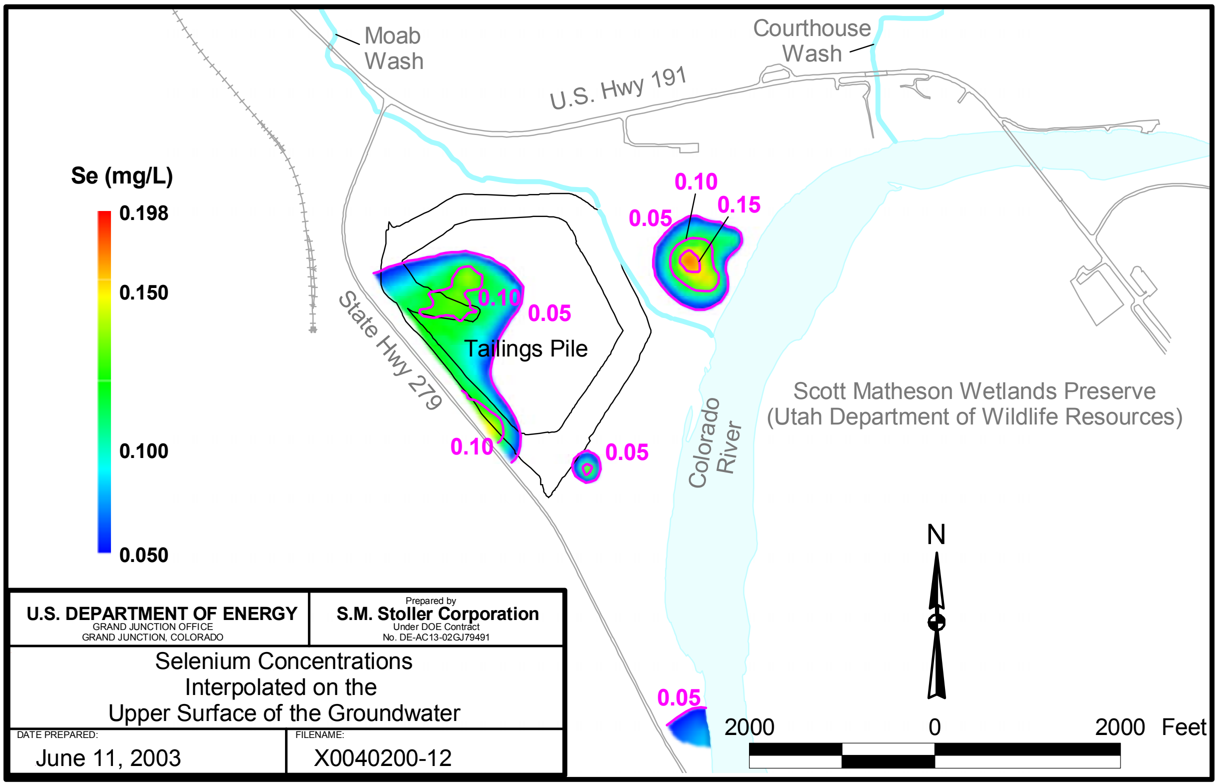

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Figure 5–38. Selenium Concentrations Interpolated on the Ground Water Surface

Elevated selenium concentrations in the west plume are associated with six locations; two monitor wells completed in the shallow ground water beneath the tailings pile (AR4D and MOA-437), two monitor wells installed along the west edge of the tailings pile in the shallow aquifer (MOA-440 and -442), and two monitor wells installed in the shallow aquifer near the toe of the pile (MW-1-R and SMI-PZ3S). Selenium concentrations exceeding the Safe Drinking Water Act standard in these wells range from 0.05 mg/L to 0.2 mg/L.

5.5.1.9 Uranium

The areal distribution of uranium concentrations greater than the 0.044 mg/L EPA standard, contoured on the surface of the ground water, is presented in Figure 5–39. Concentrations in both the tailings area and the former millsite area appear to decrease with depth in the aquifer, as evidenced by the contours interpolated at 50 ft, 100 ft, and 150 ft below the surface of the ground water and presented in Figure 5–40, Figure 5–41, and Figure 5–42 respectively.

A cross-sectional view of the uranium plume along a transect adjacent to the Colorado River is shown in Figure 5–43. Very little uranium is present at depths greater than 100 ft below the ground water surface in the millsite area and below 150 ft in the tailings pile area. The highest uranium concentrations of 17.6 and 16.5 mg/L are present in the shallow ground water at SMI-MW01 and TP-02, respectively, which are both located near the Colorado River in the former millsite area.

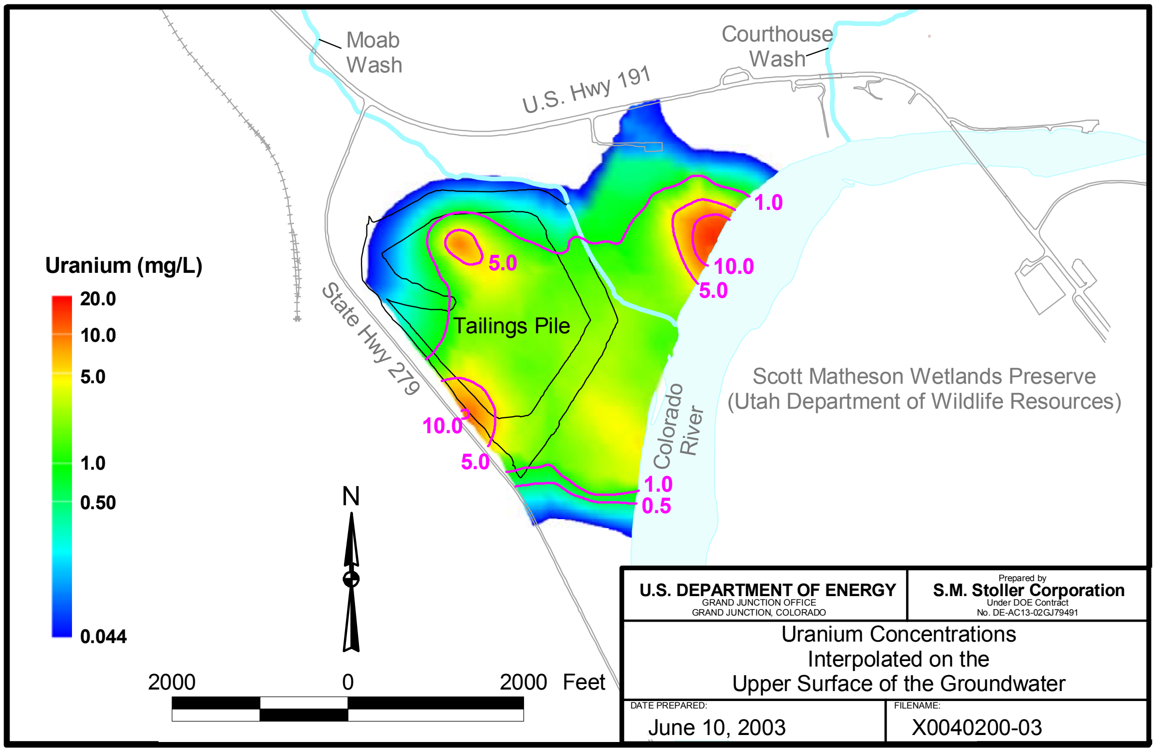

Figure 5–39. Uranium Concentrations Interpolated at the Ground Water Surface

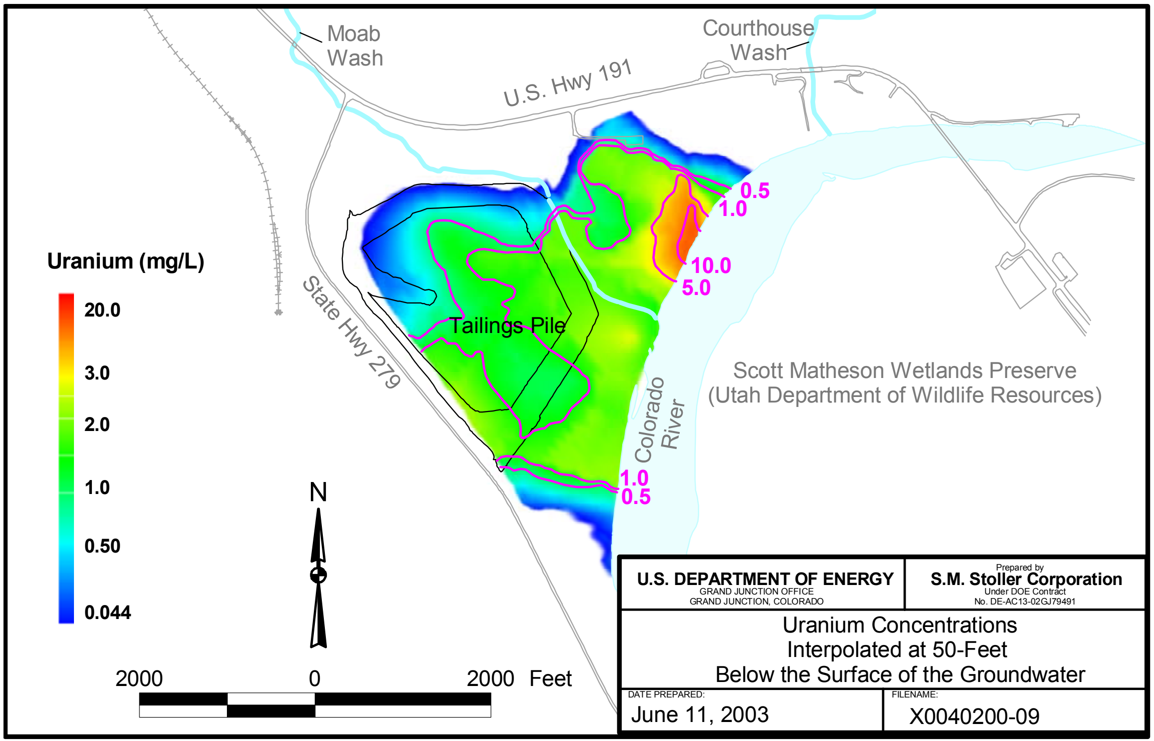

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Figure 5–40. Uranium Concentrations Interpolated at 50 ft Below the Surface of the Ground Water

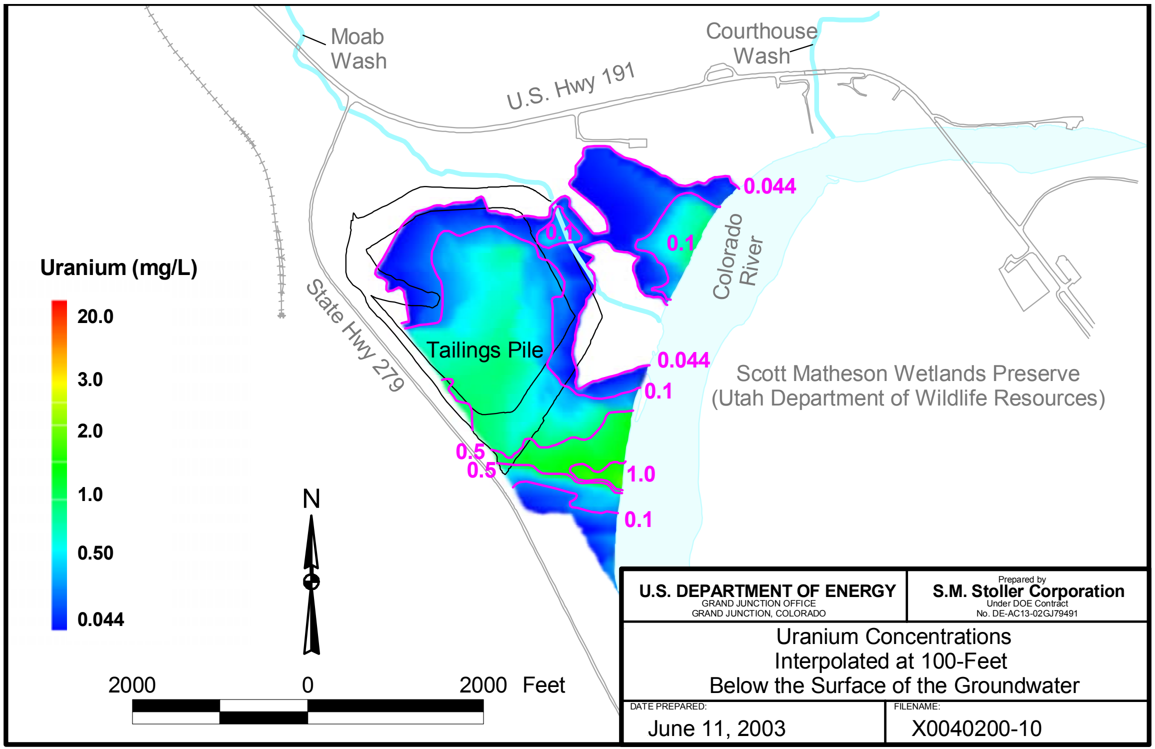

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Figure 5–41. Uranium Concentrations Interpolated at 100 ft Below the Surface of the Ground Water

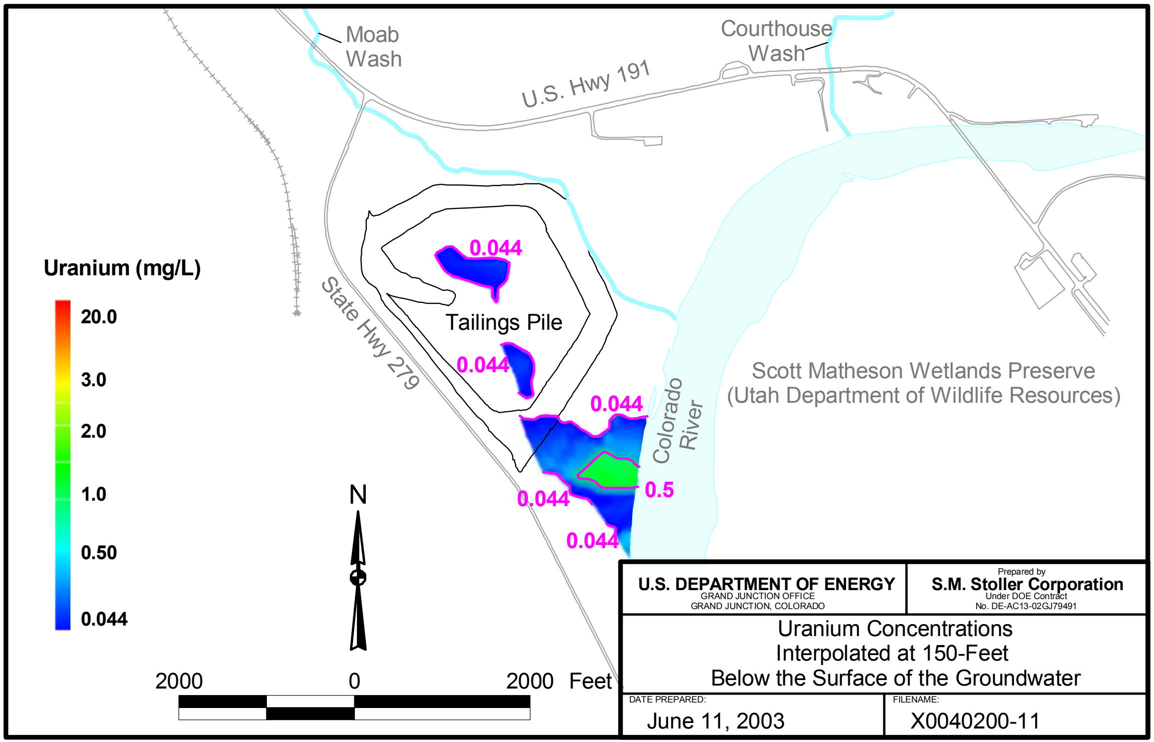

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Figure 5–42. Uranium Concentrations Interpolated at 150 ft Below the Surface of the Ground Water

In addition to the high uranium concentration, the millsite area has relatively low ammonia concentrations (Figure 5–29 in contrast to the plume downgradient of the tailings pile. SMI (2001) suggested that the high uranium concentrations beneath the millsite are caused by waste leaking from the former wood chip disposal areas. Although the uranium plume is in an area where wood chip disposal was likely to have occurred, lithologic logs of borings installed in this area of the site do not indicate that they penetrated through the wood chip pits. Another possible source of the high uranium concentrations is the uranium ore stockpiles; however, monitor wells nearest the largest known ore stockpiles have lower uranium concentrations. The plumes at the Moab site are similar to those at the Shiprock, New Mexico, UMTRA Project site, where there is also a plume near the millsite that contains dominantly uranium, and another plume that is dominated by nitrate or ammonia (DOE 2000).

Whether the source of the high uranium is the wood chip pits, the ore stock piles, or some other millsite-related release, it seems that some of the ground water contamination originates in the millsite area, independently of the tailings pile.

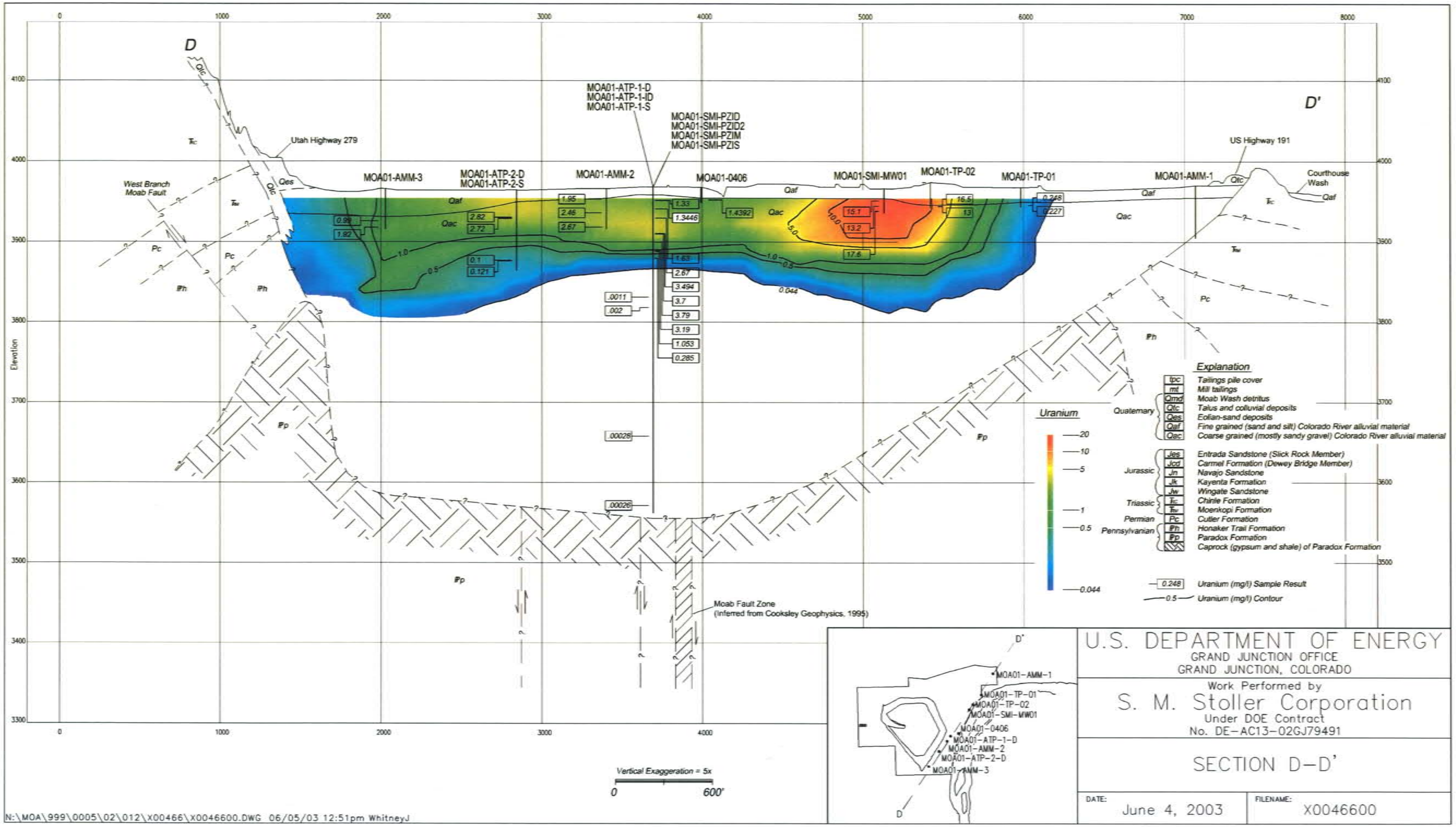

Figure 5–43. Cross-Sectional View D–D' of Uranium Concentrations in Ground Water Adjacent to the Colorado River

No text for this page

Uranium Concentration Trends

Uranium is one of the site-related constituents that have the greatest amount of historical ground water data. Most of the historical uranium data is associated with monitor wells established by Atlas Minerals to demonstrate compliance as part of their NRC operating license. ATP-2-S and ATP-2-D are former compliance wells located at the downgradient toe of the tailings pile and have been sampled for uranium since 1982. Uranium concentrations measured in ground water samples collected at these two wells are presented in the time-concentration plots in Figure 5–44 and Figure 5–45. Ground water samples from ATP-2-S were collected from a screened interval approximately 29 to 39 ft below ground surface. ATP-2-D samples were collected from a screened interval approximately 80 to 95 ft below ground surface. Sample results from both wells indicate uranium concentrations are steadily decreasing with time since 1985.

5.5.2 Organic Contaminants

In addition to the tailings, soils in the millsite area may also contain contamination that could continue to leach into the ground water system. Contamination could occur from the wood chip disposal areas or from unintended spills or leaks during mill operations (Figure 3–3). During radiometric surveys, field crews noted some petroleum stain or odor on soils in the portion of the millsite delineated on Figure 5–46. To help evaluate the likelihood of potential organic contamination in the millsite area, soil samples from six locations in the millsite area were collected in May 2002. In addition, organic analyses were performed on ground water samples collected from six wells in the millsite area, pore water samples collected from three wells completed in the tailings, and ground water samples collected from one alluvial well located downgradient near the toe of the tailings pile.

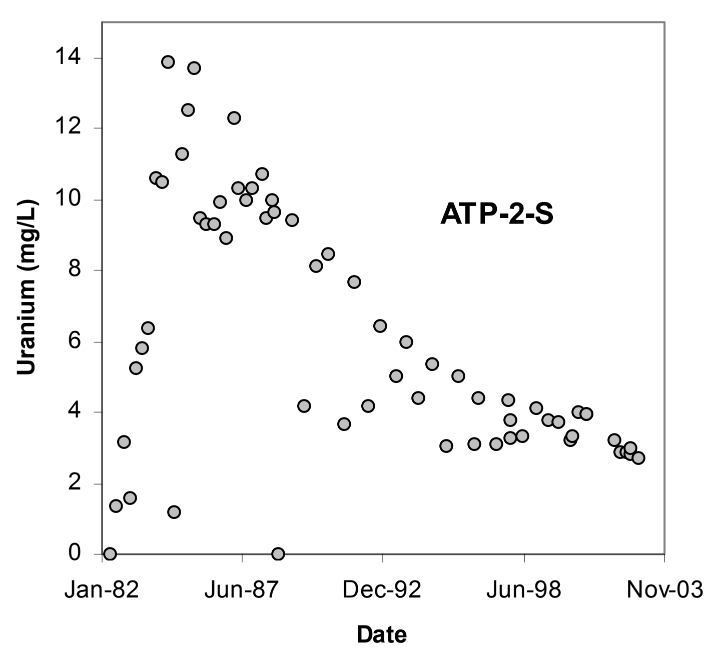

Figure 5–44. Uranium Time-Concentration Plot for ATP-2-S Located at the Downgradient Toe of the Tailings Pile

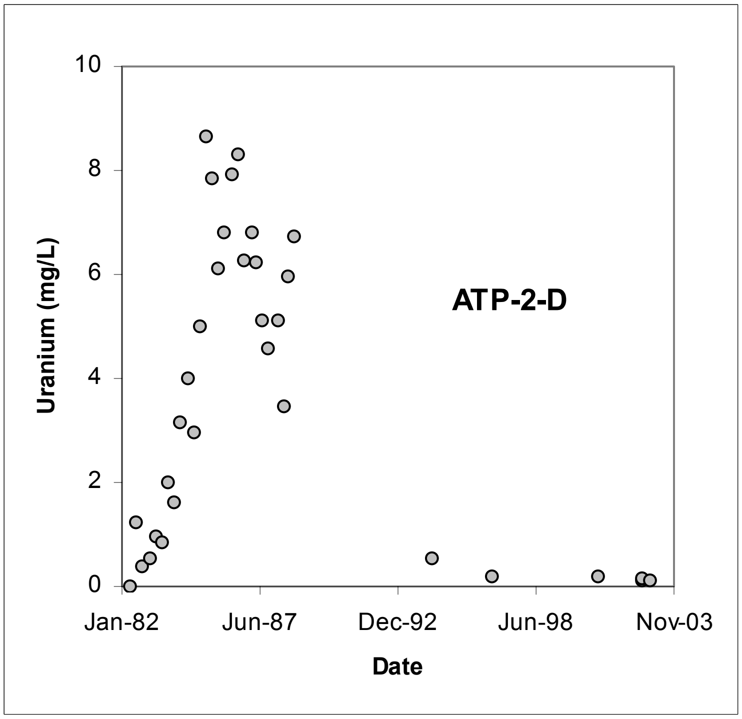

Figure 5–45. Uranium Time-Concentration Plot for ATP-2-D Located at the Downgradient Toe of the Tailings Pile

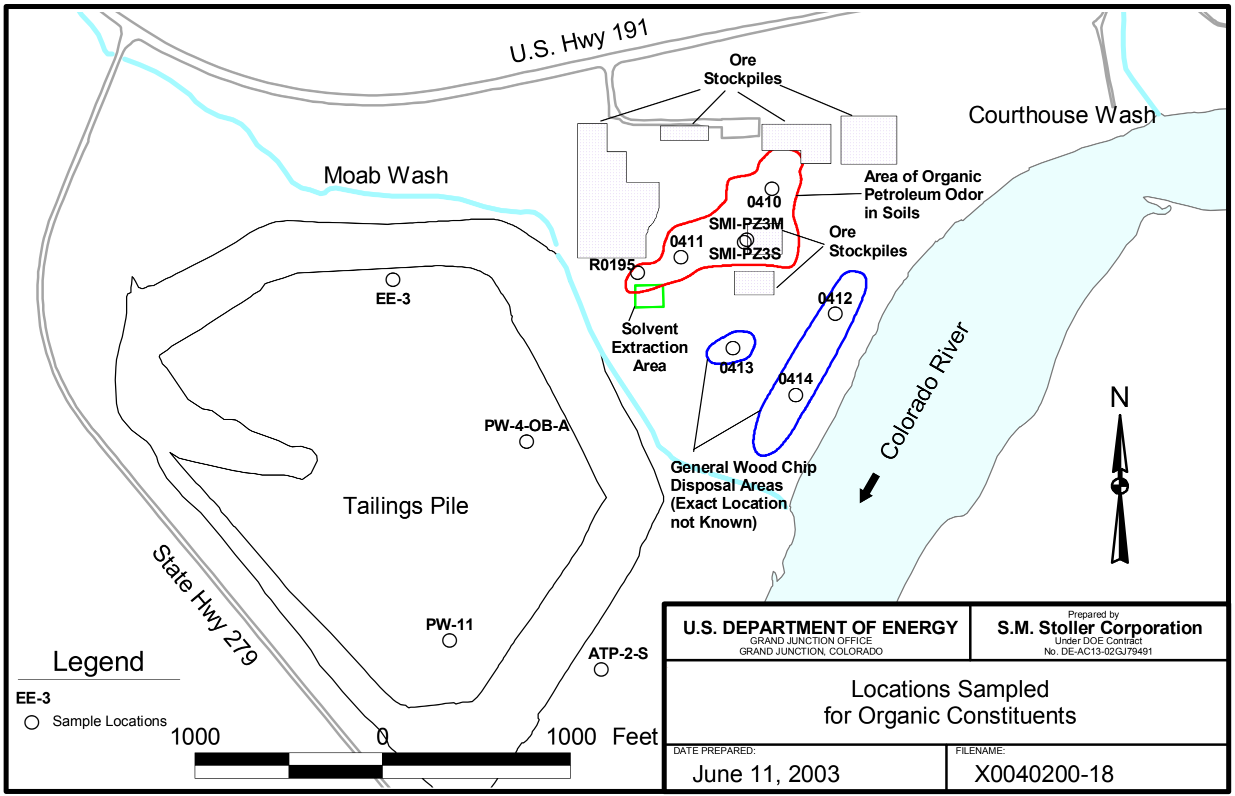

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Figure 5–46. Sample Locations Analyzed for Organic Constituents

5.5.2.1 Results of Soil Analysis

Soil samples were collected from a depth of 0 to 1 ft at temporary monitor well locations MOA-410, -411, -412, -413, -414 and soil boring location R0195 on May 23, 2002 (Figure 5–46). A duplicate sample at MOA-412 was also collected. Sample R0195 was collected from an area where the strongest petroleum odor was identified during an extensive radiometric survey. All samples were analyzed for volatile organic constituents (EPA method 8260B), selected metals (digestion by EPA method 3050), and selected radionuclides.

All organic compound concentrations were less than detection limits except for 10 compounds in sample R0195 (Table 5–22). The compounds in sample R0195 have a signature that is probably related to petroleum-based fuels. The signature is likely from a spill of either fuel-oil-based solvents used in milling or a spill from a motor vehicle. Based on the proximity to the solvent extraction facility, the contaminants at sample location R0195 are likely related to releases from this part of the milling process.

Table 5–22. Organic Constituents Detected in Soil Samples From the Moab Millsite (Results that were less than the detection limit are not shown.)

Location	Analyte	Concentration (mg/kg)	Lab Qualifiers
R0195	2,3-dimethyl napthalene	0.0802	NJ
R0195	2-butyl-1,1,3-trimethyl-cyclohexane	0.0621	NJ
R0195	3-tetradecene, (Z)-	0.0828	NJ
R0195	cyclohexane, 1,1,3,5-tetramethyl-, cis-	0.0915	NJ
R0195	unknown	0.0657	J
R0195	unknown	0.0738	J
R0195	unknown	0.0594	J
R0195	unknown hydrocarbon	10.8887	J
R0195	dodecane	0.1609	NJ
R0195	unknown	0.0733	NJ

Lab Qualifiers: N = tentatively identified compound, J = estimated

The limited organic signature at the Moab site is consistent with the infrequent detection of organic compounds at UMTRA Project sites (DOE 1997a). Eleven UMTRA sites were sampled to detect the presence of organic constituents, and of these only one (New Rifle, Colorado) had any visible petroleum stains or odor in the millsite soils. Of the organic constituents detected at the Moab site (Table 5–22), 2-butyl-1,1,3-trimethylcyclohexane, dodecane, and unknown hydrocarbons were also detected at the New Rifle site (DOE 1997b). The minor amounts of organic constituents present at the Moab site are likely to degrade and are not likely to constitute a continuing source of contamination to the ground water.

Concentrations of selected radionuclides in the soil samples are provided in Table 5–23. Maximum concentration of Ra-226 is 29.62 pCi/g, and Ra-228 concentration is less than the detection limit in all but one sample. Uranium-234 and U-238 are nearly in isotopic equilibrium and have concentrations that are reasonably consistent with chemical uranium values. Thorium-230 is not in secular equilibrium with the uranium isotopes, indicating that it was chemically separated during the milling process.

Table 5–23. Selected Radionuclide Constituent Concentrations (pCi/g) in Atlas Millsite Soils Sampled 5/23/2002

ID	Ra-226	Ra-228	U-234	U-238	Th-239
410	3.68B	0.81U	15.8E	20	32.6
411	21.31B	1.19U	66.6E	76.8	5.7
412	25.45B	1.25U	8.7E	9.5	2.2
412 Dup	3.21B	0.72U	35.2E	38.8	24
413	29.62B	1.09U	8.3E	9.4	1.7B
414	5.35B	0.91U	15.3E	17.7	2.6
R0195	1.65B	0.67	63.1E	72.1	38
R0195 Dup	1.02B				3.1
R0195 Dup					291

Lab Qualifiers:
B= Result is between the instrument detection limit and the contract required detection limit
E = Estimated value because of interference
U = Analytical result below the detection limit
Dup = duplicate

Concentrations of selected chemicals in the soils are provided in Table 5–24. Although no upgradient background soils were analyzed, uranium concentrations in the millsite soils are significantly higher than concentrations in alluvium samples collected downgradient of the Moab site (Table 5–16). The mean uranium concentration in the alluvium is 2 mg/kg, and mean crustal abundance is 1.8 mg/kg. Concentrations of ammonia, arsenic, copper, lead, molybdenum, selenium, uranium, and vanadium are highest in sample R0195, suggesting that the petroleum staining in this sample is related to solvents used in milling. Variable concentrations in the duplicates of samples MOA-412 and R0195 indicate that the soils have significant small-scale heterogeneity.

Table 5–24. Selected Metals and Other Constituent Concentrations (mg/kg) in Moab Millsite Soils Sampled 5/23/2002

ID	As	Ba	Cu	Fe	Mn	Mo	NH_3-N	Pb	Se	U	V
410	4.4	117	39.7	5,780	270	1.2	7.5	15.2	1.1*N	46.8	96
411	1.4	51.6*	5.9	4,290	230	0.9	1.1	2EN	0.38B*N	214	83.1
412	3.5	136*	9.1	8,910	322	1.5	5.9	10EN	1.2*N	21.7	20.7
412 Dup	4.9	269*	18.1	7,410	303	5	4.3	5.7EN	1.0*N	157	115
413	3.2	148*	9.1	8,000	318	1.7	6.9	10EN	1.1*N	22.8	18.8
414	2.2	93.1*	6	6,120	433	0.2B	5.7	3EN	1.1*N	49.7	88.3
R0195	12.9	141*	174	8,150	287	6.3	56.8	31EN	6.4*N	45.2	1,330
R0195 Dup										32	
R0195 Dup										204	

Lab Qualifiers:
* = Replicate analysis not within control limits
B= Result is between the instrument detection limit and contract required detection limit
E = Estimated value because of interference
N = Spike recovery not within control limits
Dup = duplicate

5.5.2.2 Concentrations of Organic Constituents in Ground Water

Ground water was sampled on May 23 and May 30, 2002, for an investigation of potential organic contamination. Concentrations of pesticides, semivolatile organic compounds, and volatile organic compounds were analyzed in ground water samples collected at locations

MOA-410, -412 (in duplicate), -413, -414, SMI-PZ3M, and SMIPZ3S in the millsite area (Figure 5–46). The same organic compounds were analyzed in tailings pore water samples collected at locations PW-11 and PW-4-OB-A (Figure 5–46). Volatile organic compounds were also analyzed in a sample of tailings pore fluid collected from well EE-3 and in a sample from alluvial aquifer well ATP-2-S located downgradient of the tailings pile; concentrations of all volatile organic compounds in these two samples were less than the detection limits. Concentrations of pesticides were less than detection limits in all samples.

A summary of the organic constituents detected in the ground water and pore water samples is presented in Table 5–25. Most of the compounds listed in Table 5–25 could be related to the milling operation. Phlalates are plasticizers; for example, bis(2-ethyhexyl)phlalate is often used in pumps such as those used at Atlas. Phosphate-based compounds could be related to the solvent used in the solvent extraction process or process resins. Several of the compounds are related to fuels such as the kerosene, which was used as a carrier for the phosphate-based solvents. Only a few organic compounds were detected in the ground water, and most concentrations are low. It appears that organic contamination in ground water is not widespread and is limited to low concentrations of a limited number of compounds. This is consistent with findings at other uranium milling sites in the UMTRA Project (DOE 1997a). The dissolved organic compounds will likely biodegrade over a reasonable time period.

Table 5–25. Organic Constituents Detected in Ground Water and Pore Water Samples From the Moab Millsite (Results that were less than the detection limit are not shown.)

Analyte	Frequency of Detection	Maximum Concentration (ug/L)	Well with Maximum Concentration
Tailings Pore Fluid			
bis(2-ethylhexyl)phlalate	2/2	19.9	PW-11
phosphine oxide, triphenyl-	1/2	62.5	PW-4-OB-A
1,3-isobenzofurandione, 4,7-dimethyl-	1/2	17.5 NJ	PW-11
1,8-naphthalic anhydride	1/2	9.9 NJ	PW-11
phosphoric acid, tributyl ester	2/2	224.2 NJ	PW-4-OB-A
2-fluoro-6-nitrophenol	1/2	7.1 NJ	PW-4-OB-A
unknowns	2/2	62.4 J	PW-11
unknown hydrocarbons	2/2	45.6 J	PW-11
unknown phthalates	2/2	13.1 J	PW-4-OB-A
Millsite Ground Water			
bis(2-ethylhexyl)phlalate	6/7	18.7	SMI-PZ3M
bromomethane	1/7	8 J	SMI-PZ3M
chloromethane	1/7	9 J	SMI-PZ3M
phosphine oxide, triphenyl-	4/7	13.6 NJB	SMI-PZ3S
dodecanoic acid, 1,2,3-propanetriyl ester	1/7	8 NJ	SMI-PZ3S
unknowns	3/7	66.2 J	SMI-PZ3M
unknown phthalates	2/7	10.3 J	SMI-PZ3S

Lab Qualifiers:
B= Result is between the instrument detection limit and the contract required detection limit.
N = tentatively identified compound
J = estimated

5.5.3 Fate and Mobility of Ground Water Contaminants

As discussed in Section 5.5.1, the most prevalent constituents in ground water at the Moab site are inorganic constituents that were associated with the processed ores or process chemicals.

Although there are similarities in the distribution of some constituents in the ground water, there are noticeable differences among most, which can be attributable to differences in source or differences in the various physical and chemical processes that govern their behavior in the subsurface.

As described in previous sections, the tailings pile at the Moab site has served as a major source of ground water contamination. The pile will continue to affect ground water as long as it is in place and pore fluids continue to leach and transport constituents to ground water. On the basis of the distribution of contaminants in ground water, it appears that two distinct areas of contamination are associated with the tailings pile: (1) a relatively deep plume that is probably associated with effluent disposed of in the pond during early operations and (2) a shallower plume located at the toe of the pile and probably a result of seepage from the former sump/French drain system located there. The most pervasive constituents located in these areas associated with the pile are ammonia and sulfate, which extend to significant depths below the pile. Some metals, such as uranium and molybdenum, are also found in association with the pile, but the highest concentrations of these constituents are typically found at shallower depths in the aquifer. Though uranium and molybdenum are considered to be relatively mobile metals, their much lower concentrations in tailings fluids relative to ammonia and sulfate may have caused a preferential partitioning of these constituents at shallower levels due to interaction with aquifer materials. Restriction of normally mobile metals to less saline portions of the aquifer could also be related to a “salting out” process that has been identified in estuaries influenced by seawater (Turner et al. 2002). In this study it was found that distribution coefficients (K_d) for certain metals increased with increasing salinity. Constituents with typically lesser mobility, such as selenium, arsenic, cadmium, and vanadium, have a much more restricted distribution, located mainly in the tailings fluids or at the immediate toe of the pile.

One area of contaminated ground water that appears unrelated to contamination directly from the tailings pile is on the mill processing area in the vicinity of the wood chips disposal area. This location contains elevated concentrations of arsenic, selenium, and vanadium, as well as some of the highest uranium concentrations associated with the site. Ammonia and sulfate concentrations in this vicinity are relatively low. Contamination appears to be relatively shallow and is quite restricted, except for uranium, probably because of the low mobility of these constituents. This contamination seems to have more of an ore-related signature, as opposed to the tailings pile contamination that is dominated by processing chemicals. Ore was stored in numerous open stockpiles in this part of the site, and it is possible that contamination could have been from leaching of those piles. There is, however, anecdotal evidence that uranium-laden fluids were disposed of in a “catch pit” in that vicinity (ORNL 1998a).

The distribution of most site-related constituents in the ground water at the Moab site has probably been the result of a variety of complex and interacting processes that have likely changed over time as processing chemistry changed. Adsorption, precipitation, and dissolution, among other processes have generated the contaminant distribution that exists today. As noted in Section 4.7, collectively a number of these processes are often modeled using a bulk transport parameter (distribution coefficient) that describes the retardation of trace constituents in the ground water system. Site-specific measurements of R_d were determined for uranium and ammonia at the Moab site. Similar measurements were made for other constituents of interest at a number of other UMTRA Project sites. Contamination patterns for most trace constituents at the Moab site are consistent with results of other R_d measurements completed to date and can reasonably be explained with the simplified bulk transport model. With respect to metals, uranium and molybdenum tend to be most mobile and more widely distributed. Other constituents such as arsenic, selenium, vanadium, and cadmium are less mobile and more restricted.

Constituents that are present in large concentrations, such as ammonia and sulfate, may behave in a more complicated fashion than predicted by a simple R_d model. This is particularly true of ammonia, which can undergo a number of transformations. The term "ammonia" is used throughout this document to indicate the total amount of ammonia species, including both the uncharged (NH_3^0) and ammonium ion (NH_4^+). The dominant dissolved ammonia species in the ground water contamination plume is ammonium ion (NH_4^+) (Section 4.10). Due to the high sulfate concentration, significant ammonium sulfate ($NH_4SO_4^-$) is also present. Under oxidizing conditions, ammonia reacts to form nitrite (NO_2^-), nitrate (NO_3^-), or nitrogen gas (N_2). Some of the transformation reactions are catalyzed by microbiological activity. Ammonia (mainly) and nitrate were used during the milling process at the Moab site. Some of the nitrate in the ground water may be an oxidation product of ammonia. The similarity in configuration of the ammonia and nitrate plumes at the Moab site indicate that this oxidation probably is occurring and can result in natural degradation of the ammonia plume over time.

Ammonia, sulfate, and other constituents probably became concentrated near the top of the tailings pile as mill pond fluid evaporated and precipitated as salts in the upper portion of the pile. The precipitation of gypsum ($CaSO_4$) can partition significant amounts of sulfate into the solid phase. Much of the sulfate concentration gradient in ground water is caused by mixing with other ground water and dispersion.

5.6 Surface Water Quality

The primary surface water body of concern for the Moab site is the Colorado River, which forms the site's eastern boundary. The Colorado River Basin Water Quality Control Project was established in 1960 (HEW 1961) and much of the early monitoring of the river was carried out in support of this project. Effects of uranium milling, especially erosion of and seepage from the uranium mill tailings piles, on river water quality was of particular concern (HEW 1966). In the 1970s, much of the focus on the Colorado River Basin concerned salinity control, pursuant to passage of the Colorado River Basin Salinity Control Act (Public Law 93-320; June 24, 1974). More recently, surface water monitoring of the Colorado River Basin has been carried out as required by the Clean Water Act.

Site-specific surface water monitoring was historically conducted by the Atlas Corporation and essentially consisted of an upstream and downstream location, outside of the mixing zone of ground water discharge to the river. More intensive and rigorous sampling and monitoring began in the mid-1990s, as Atlas sought approval for remediation and closure plans for the mill. Subsequent to Atlas's bankruptcy, the Moab Reclamation Trust initiated monitoring activities to support site reclamation plans. DOE completed several rounds of surface water monitoring after assumption of responsibility for the Moab site.

Section 3.6.2 presents a more detailed discussion of historical monitoring at the site. Pertinent results of those studies along with more recent monitoring results are presented in this section. Regional surface water quality is described first, followed by site-specific monitoring results.

5.6.1 Regional Conditions

The Moab area is located within the Southeast Colorado Watershed Management Unit as designated by the Utah Department of Environmental Quality's Division of Water Quality (UDEQ 2000). This watershed unit includes the Colorado River in the vicinity of the Moab site as well as all of its tributaries and other water bodies between the Colorado River and the Colorado/Utah state line. The major tributary to the Colorado River upstream of the Moab site is the Dolores River, which enters the Colorado River about 32 miles from the site. Lesser tributaries include Onion Creek, Castle Creek, and Mill Creek. Water also enters the river from Courthouse and Moab washes following precipitation events.

Waters in the Southeast Colorado Watershed Management Unit have been monitored on a regular basis to evaluate long-term water quality trends. An intensive monitoring program took place between July 1997 and June 1998 to assess streams against state water quality standards and pollution indicators to determine if their designated beneficial uses were being met (UDEQ 2000). The Colorado River and its tributaries from Lake Powell to the Utah-Colorado state line have the following designated uses:

- 1C—source of drinking water.
- 2B—contact recreation.
- 3B—warm water game fish.
- 4—agricultural use, including irrigation and stock watering.

Some of the earliest studies of Colorado River water quality were undertaken mainly to study the effects, if any, of uranium milling operations on the river. Radionuclides, particularly radium, were of main concern in these studies (HEW 1966). A study conducted to determine the potential effects of the Moab site identified several constituents in the Colorado River in the study area that had concentrations above recommended limits, including sulfate, chloride, TDS, and manganese (HEW 1961). The presence of these constituents was attributed to natural causes. Highest levels of some analytes were detected at the confluence of the Dolores River with the Colorado.

Salinity control efforts began in the Colorado River Basin in the early 1970s. Salinity of the Colorado River generally increases progressively downstream because of lower quality ground water inflow and return flows from surface irrigation. A major source of salinity load to the Colorado River, particularly in the Southeast Colorado Watershed Management Unit, is the Dolores River. As the Dolores River crosses Paradox Valley in southwestern Colorado, highly saline ground water (brine) discharges to the river (Chafin 2003). The source of the brine is a collapsed salt anticline, similar to that in Moab valley. Near its confluence with the Colorado River, the salinity of the Dolores River limits the use of river water for irrigation of some crops (UDEQ 2000).

Onion Creek is another high-salinity tributary to the Colorado River. Onion Creek Spring is fed by ground water that leaches salts from the underlying Paradox Formation. The spring discharges to Onion Creek about 6 miles above its confluence with the Colorado River. Onion Creek has been designated as an impaired waterbody due to elevated levels of TDS from both natural and agricultural sources (UDEQ 2000). Several other waterbodies in the Southeast Colorado Watershed Management Unit have been designated as impaired because of high TDS levels, including Mill Creek, which is a source of recharge to the alluvial aquifer across the Colorado River from the Moab site.

On a regional basis, selenium concentrations in the Colorado River have also been of concern. Upstream of the Moab area, the Colorado River passes through large areas of exposed Mancos shale, a geologic unit that has high concentrations of selenium (Seiler et al. 1999). Runoff from the Mancos and irrigation return flows from unlined canals crossing the Mancos contribute to selenium concentrations in the river. Although selenium levels have received greater attention in the Upper Colorado River Basin in Colorado, where concentrations have been detected up to two orders of magnitude above the AWQC of 0.005 mg/L (Spahr et al. 2000), concentrations in the vicinity of the Moab site are also relatively high (see Section 5.5.2). Other constituents are known to be elevated in the Upper Colorado River Basin and the Southeast Colorado Watershed Management Unit as a result of the extractive industries; these effects tend to be more localized (Spahr et al. 2000; UDEQ 2000).

Despite the different factors that define the surface water quality of the Colorado River, the portion of the river belonging the Southeast Colorado Watershed Management Unit was assessed as fully supporting all of its beneficial uses according to results of intensive monitoring conducted from July 1997 to June 1998. Therefore, the overall river water quality is considered to be good. However, of the 981 stream miles contained within the Southeast Colorado Watershed Management Unit, only 27 sampling sites were used in the assessment. Four of the 27 sampling sites were located on the Colorado River itself (UDEQ 2000). As discussed below, the Moab site has been demonstrated to cause localized degradation of surface water quality (see Section 5.6.5).

5.6.2 Site Background Characterization

Section C.1 in Appendix C describes the process used to develop an initial analyte list for surface water. A baseline round of sampling was completed in summer 2002, and results of that sampling were used to reevaluate surface water data and make recommendations for inclusion/exclusion of constituents for risk assessment or future monitoring. The baseline sampling included major ions and a comprehensive list of potential site-related constituents based on their detection in ground water and/or tailings fluids or based on historical process knowledge.

The following constituents were detected at background location CR-1, which was used for the baseline background location: aluminum, ammonia, arsenic (very low), barium, boron, calcium, chloride, fluoride, gross alpha, gross beta, iron (unfiltered only), lithium, magnesium, manganese, molybdenum (very low), nickel (very low), nitrate, polonium-210, potassium, radium-226 (low), selenium, sodium, strontium, sulfate, TDS, uranium, vanadium, and zinc (very low). Constituents that were analyzed but not detected included antimony, beryllium, bismuth, cadmium, chromium, cobalt, copper, elemental lead, lead-210, mercury, radium-228, radon-222, silver, thallium, thorium-230, phosphate, and tungsten.

Detectable constituents and concentration ranges at background locations for samples collected during SMI and GJO sampling events from April 2000 through December 2002 are presented in Table 5–26. Background locations include CR-1 and CR1COMP. USGS collected background surface water data from August 1998 through February 2000. Those data are not included in Table 5–26 but are discussed in the text.

Table 5–26. Range of Constituents Detected in Background Surface Water

Constituent	Frequency of Detection	Range (mg/L except as noted)
Major Ions		
Calcium	16/16	46.3–141
Chloride	20/20	25.1–172
Magnesium	16/16	12.9–41
Potassium	16/16	2.1–5.3
Sodium	17/17	30.5–125
Sulfate	20/20	84.1–439
Total Dissolved Solids	12/12	430–1060
Minor Constituents		
Aluminum	9/12	0.008–0.14
Ammonia, total as N	9/20	Nd–0.134
Arsenic	8/11	Nd–0.002
Barium	13/13	0.051–0.14
Boron	4/10	Nd–0.123
Copper	3/13	Nd–0.0014
Fluoride	3/3	0.3–0.504
Gross Alpha	1/7	Nd–13.82*pCi/L
Gross Beta	2/7	Nd–13.78**pCi/L
Iron	6/9	Nd–4.17**
Lithium	1/3	Nd–0.0557
Manganese	8/18	Nd–0.076
Molybdenum	17/18	Nd–0.007
Nickel	7/10	Nd–0.002
Nitrate as NO_3	6/6	0.776–5.51
Polonium-210	2/5	Nd–0.1142 pCi/L
Radium-226	5/5	0.12–0.23 pCi/L
Selenium	15/15	0.0013–0.0079
Strontium	10/10	0.965–1.63
Uranium	20/20	0.0023–0.008
Vanadium	11/11	0.0007–0.0031
Zinc	5/12	Nd–0.006

*detected in only one unfiltered sample
**highest value is for an unfiltered sample
Nd = not detected

Location CR-1, although used as a baseline for evaluating surface water quality, may not represent the range of true background water quality for the site. This location is upstream of the influence of the Paradox Formation, which significantly affects ground water in the site area (Figure 5–18). Ground water discharging to the Colorado River in the vicinity of location CR-1 is therefore significantly different from ground water that discharges farther downstream. Studies conducted on the Dolores River to determine the effect the salt anticline beneath Paradox Valley has on water quality have demonstrated considerable differences in river water quality upstream and downstream of the valley (Chafin 2003; USGS 2000). Dissolved solids, sodium, and chloride all increased downstream of the Paradox Valley compared to upstream concentrations. Although the emphasis in those studies was on those major indicators of water quality, it should be expected that other minor constituents could be affected as well.

5.6.3 Distribution of Major Constituents in Surface Water

A majority of surface water samples collected in the 2000–2002 time frame had major constituent concentrations within the range of background. However, samples from a number of locations had concentrations that were elevated above background, in some cases several orders of magnitude. Chloride and sodium were the most commonly elevated constituents in these samples, though calcium, potassium, and, to a lesser extent, magnesium, also had elevated levels. The sample locations that produced these higher concentrations included the following: 206, 207, CR2B, CR2BX, CR2BY, CR3, CR3-001, CR4, CR5, CRBB, CRBBX, CRBSEEP, CRC, CRX, Moab Wash, and Moab Seep. Though location coordinates for Moab Seep are lacking, it is assumed that this is a sample collected from a seep in the vicinity of Moab Wash.

All of the above-noted samples are from locations immediately adjacent to the riverbank, and most are from the sandbar area immediately downstream from the confluence of the Colorado River with Moab Wash, adjacent to the tailings pile. Two of the locations—CR2BX and CR2BY—were described in trip reports as being stagnant pools. Concentrations at these locations, and possibly others characterized by very slow moving water, have probably been affected by evaporation (among other processes), causing an increase in dissolved constituents. A number of the other locations were described as small channels that were separated from the main channel of the Colorado River by alluvial deposits. Samples from most of the locations that were monitored over time had considerable variation in analyte concentrations; at some times concentrations in those locations were within the range of background. Several factors probably contribute to the chemical variation in the samples, the most important of which is river discharge rates. River flow conditions influence the degree to which ground water discharge can influence surface water chemistry. There is some indication that the Colorado River is a losing stream that recharges the ground water system during high flow periods, and under low flow conditions ground water discharges to the river, particularly in backwater areas created by lowered river water levels. Ground water-surface water interactions and consequences of this relationship are discussed in more detail in Section 5.6.6. and 5.6.8.

As noted in the discussion of regional surface water quality, some surface waters in the Moab area are influenced by discharge of saline ground water containing dissolved salts from the Paradox Formation that is found in the cores of salt anticlines that are characteristic of the region (see Section 5.6.1). Highly saline ground water is known to occur beneath the Moab site as well as at the Matheson Wetlands Preserve across the river from the site. Because ground water gradients on both sides flow toward the river, it is likely that the presence of the brine should have some effect on surface water quality. However, because process fluids disposed of in the former tailings pond contained some of the same constituents that occur in natural brines, distinguishing between naturally occurring constituents and site-related constituents in surface water is not straightforward. Increases in sodium, chloride, or dissolved solids content of river water (among other constituents) in the vicinity of the site, compared to background concentrations at location CR-1, could be a result of discharge of either site-related contaminated ground water or natural brines. Sulfate, though considered a major ion, is discussed in Section 5.6.4.

5.6.4 Distribution of Minor Constituents in Surface Water

During the baseline sampling event, river samples were collected from locations adjacent to the former milling area and adjacent to the mill tailings pile. Samples were analyzed for the same baseline analytes as discussed in Appendix C. A number of constituents were either not detected or did not significantly exceed concentrations observed for background locations. These constituents were aluminum, antimony, arsenic, barium, cadmium, chromium, cobalt, copper, nickel, radium-226, radium-228, elemental lead, lead-210, polonium-210, thorium-230, thallium, silver, vanadium, and zinc. With the exception of barium and copper, these constituents are not discussed further in this section.

For comparison to surface water quality data, Table 5–27 provides benchmarks for protection of aquatic life for selected constituents detected in Colorado River water. Of those constituents listed in Table 5–27, only ammonia and selenium have established federal water quality criteria. Other benchmarks were obtained from the literature for the protection of freshwater aquatic life.

Table 5–27. Surface Water Benchmarks for Protection of Aquatic Life

Constituent	Benchmark (mg/L)	Reference
Ammonia	varies	See discussion in Section 5.5.5
Barium	0.0039 (SCV[a]), 0.11 (SAV[b])	Suter and Tsao 1996
Boron	0.0016 (SCV)	Efroymson et al. 1997
Chloride	230 (chronic) 860 (acute)	EPA 1998
Manganese	0.12 (SCV), 2.3 (SAV)	Suter and Tsao 1996
Molybdenum	0.37 (SCV) ; 16 (SAV)	Suter and Tsao 1996
Selenium	0.005	NAWQC[c], EPA 2002
Strontium	1.5 (SCV), 15 (SAV)	Suter and Tsao 1996
Sulfate	100 (acute/chronic—fish)	Haines et al. 1994
Uranium	0.0026 (SCV) , 0.046 (SAV)	Suter and Tsao 1996

[a] Secondary Chronic Value
[b] Secondary Acute Value
[c] National Ambient Water Quality Criteria

Comparison of the benchmarks in Table 5–27 with background concentration ranges provided in Table 5–26 indicates that background levels of barium, boron, selenium, strontium, and sulfate exceed some of the benchmarks. Thus, the exceedence of these benchmarks at locations adjacent to and downgradient of the Moab site is not necessarily an indication of site-related contamination. Appendix C−1 provides a listing of locations where benchmarks were exceeded, along with concentrations and any comments regarding sample location. It can be seen that many of the locations where constituent concentrations exceed benchmarks are the same ones where concentrations of major constituents exceed background ranges. Again, these locations are mainly located immediately adjacent to the riverbank. As noted in descriptions included for some samples, many of these samples were collected in shallow water with little or no surface water flow.

A screening of surface water contaminants was completed during preparation of the Biological Assessment for the Moab site (Appendix A of DOE 2003b) and included an evaluation against background concentrations as well as benchmarks. That analysis resulted in the identification of five constituents as contaminants of potential concern in surface water. Those constituents were ammonia, uranium, sulfate, manganese, and copper. A more detailed discussion of the distribution and concentrations of those constituents in surface water is provided in Section 5.6.5.

Two recent studies looked at distributions of constituents at transects across the river (HLA 1998, SMI 2001). Results of those efforts can be used to make several generalizations about distribution of constituents in surface water compared to background concentrations. As noted above, concentrations of most constituents exceed background ranges only in samples that are close to the riverbank. With increasing distance from the shoreline, most constituents are within ranges observed in background. Again, there may be multiple causes of elevated nearshore concentrations. However, the influence of these processes appears to diminish rapidly with distance into the main channel of the river. Table 5–28 presents data for two locations along the river; one set of data for each location was collected from immediately adjacent to the shore and another was collected 1 ft farther into the river (locations with –001 designations). Location CR2B is located in the sandbar area adjacent to the tailings pile. Water in this area is generally shallow due to the presence of sandbar deposits, and flow is generally very slow; during the August 2002 sampling, a sample collected at this location was actually from a pool that had no flow. Location CR3 is downstream from CR2B. At this point the river channel is divided into two longitudinal segments separated by a large sandbar deposit. Based on descriptions for some events, water flow in this area was typically slow and depths were shallow (see Appendix C).

Table 5–28. Ranges of Selected Constituents in Surface Water at Four Locations

Constituent	**Range in Concentration (mg/L)**			
	CR2B	**CR2B-001**	**CR3**	**CR3-001**
Ammonia, total as N	0.05–235	0.335–5.652	0.18–11.335	1.033–33.230
Barium	0.0533–0.182	0.0537–0.104	0.0516–0.103	0.0545–0.119
Chloride	24–955	109–183	24–466	124–7990
Gross alpha	43–665	Nd–23.62	Nd–27.65	Nd–390
Manganese	Nd–12	0.0216–0.992	Nd–0.150	0.0154–1.590
Molybdenum	0.003–0.639	Nd–0.0146	Nd–0.0115	Nd–0.102
Strontium	1.46–7.5	1.03–1.58	1.06–1.97	1.04–10.2
Sulfate	78.5–4820	280–449	78–605	315–2880
TDS	676–8,650	748–1,080	930–1,780	810–10,800
Uranium	0.0025–1.22	0.0165–0.0358	0.0026–0.0516	0.0185–0.630

Table 5–28 shows ranges of concentrations for selected constituents at two locations where river water samples were collected at the river/bank interface and at 1 ft from the interface (2001−2002 data). The data indicate that constituent concentrations span a large range at each location. This variation may be largely controlled by differences in river discharge during different sampling events (river flows ranged from about 1,600 to more than 14,000 at the Cisco gaging station at the times that sampling occurred). However, the data in the table also indicate the importance of channel geomorphology in controlling surface water quality at a given location. At location CR2B, the high end of the concentration range for all constituents is lower for samples collected 1 ft from the shoreline. At location CR3, the opposite relationship is evident—the offshore samples span a higher concentration range. At location CR2B, alluvial deposits are located immediately adjacent to the riverbank, whereas at CR3, the deposits are located farther out in the river channel. The variation in shoreline configuration sets up highly

localized sets of flow conditions and velocities, which greatly influence surface water quality across the river channel. This is an important consideration in designing a surface water compliance monitoring strategy for evaluation of attainment with surface water quality goals.

5.6.5 Nature and Extent of Site-Related Surface Water Contamination

One of the earliest studies of the Colorado River in the vicinity of the Moab site was conducted by the U.S. Department of Health, Education, and Welfare as part of the Colorado River Basin Water Quality Control Project. This study took place when Atlas was still discharging effluent directly to the Colorado River. Samples of effluent and river water above and below the discharge point were collected (the closest downgradient point was 2 miles below the millsite). Despite the fact that very low river flows were present during the study (Colorado River flow averaged about 1,500 cfs or less during the study period), no differences were noted in chemical or radioactive constituents above and below the millsite. The greatest influence on Colorado River water quality appeared to be the Dolores River—radium, uranium, sodium, chloride, and TDS all increased in the Colorado River at the confluence with the Dolores River (compared to an upstream sampling point). This study suggests that even during its operational period, the site's influence on river water quality was limited in extent.

Selenium, barium, strontium, boron, and sulfate are constituents with background concentrations that exceed aquatic benchmarks. Barium, boron, and selenium levels are not significantly higher adjacent to the site than in background areas and are therefore assumed to be not site-related. Strontium levels are higher than background in some locations, though these are mostly the stagnant pools or slow-moving waters. Molybdenum is most likely site-related, but concentrations are elevated in relatively few surface locations—mainly pools where evaporation may influence concentrations. Because of their occurrence in background or the fact that concentrations are only infrequently elevated in isolated areas, the above constituents are not discussed further in this section. The remainder of this section will focus on the contaminants of potential concern identified in the Biological Assessment—ammonia, copper, manganese, sulfate, and uranium.

Ammonia

Ammonia levels in Colorado River water in some locations adjacent to and downstream of the Moab site exceed the chronic and acute criteria National Ambient Water Quality Criteria (referred in this document as the NAWQC or "federal criteria") and proposed State of Utah surface water standards (which are identical to the federal criteria) for protection of aquatic life. (Because the state of Utah standards are not yet final, this discussion will simply refer to the federal criteria.) Ammonia levels have been of concern since ammonia was first detected in the river, largely because of the designation of this segment of the Colorado River as critical habitat for the Colorado pikeminnow (previously referred to as Colorado squawfish; see further discussion in Section 5.6.8) and three other endangered fish species (razorback sucker, bonytail chub, and humpback chub). Pikeminnow favor slow-moving backwater areas of the river as nursery habitat for young-of-year fish. As with other constituents discussed above, ammonia concentrations in the river vary considerably both temporally and spatially and are highly dependent on flow conditions of the river.

The federal acute criteria for ammonia vary as a function of pH; different sets of acute criteria have been developed according to the presence or absence of salmonids (trout). The chronic criteria vary as a function of both pH and temperature; different sets of criteria have been developed for early life stage fish present and absent. For the Moab site, young-of-year fish are of concern. Because trout do not occupy the river in the vicinity of the site, standards for salmonids absent are applicable. However, even by narrowing down the sets of criteria that are appropriate at the Moab site, numerical aquatic standards for ammonia can vary greatly simply due to temperature and pH fluctuations, which can change dramatically during even a one-day period.

Federal ammonia criteria for both chronic and acute exposures are presented in Table 5–29 through Table 5–31. Values are expressed in milligrams per liter total ammonia as N. Because the standards vary greatly and are more sensitive to some pH and/or temperature ranges than others, the federal criteria documentation (EPA 1999) does not recommend using an average temperature and pH to calculate applicable standards. Rather, calculating a range of standards that may apply under observed combinations of pH and temperature gives a better indication of applicable standards and their variability. It is recommended that standards be calculated and that ratios of observed concentrations to applicable standards be used to evaluate surface water data for compliance purposes.

Table 5–29. Concentrations of Total Ammonia as N (1 hour) pH-Dependent Values of the CMC (Acute Criterion)

CMC, mg NH_3–N/L					
pH	**Salmonids Present**	**Salmonids Absent**	**pH**	**Salmonids Present**	**Salmonids Absent**
6.5	32.6	48.8	7.8	8.11	12.1
6.6	31.3	46.8	7.9	6.77	10.1
6.7	29.8	44.6	8.0	5.62	8.40
6.8	28.1	42.0	8.1	4.64	6.95
6.9	26.2	39.1	8.2	3.83	5.72
7.0	24.1	36.1	8.3	3.15	4.71
7.1	22.0	32.8	8.4	2.59	3.88
7.2	19.7	29.5	8.5	2.14	3.20
7.3	17.5	26.2	8.6	1.77	2.65
7.4	15.4	23.0	8.7	1.47	2.20
7.5	13.3	19.9	8.8	1.23	1.84
7.6	11.4	17.0	8.9	1.04	1.56
7.7	9.65	14.4	9.0	0.885	1.32

Table 5–28 shows that ammonia concentrations (as well as other constituents) at a given location can vary widely over time. Figure 5–47 and Figure 5–48 shows that at any given location ammonia standards that apply can also vary dramatically. Because of the dynamic nature of the surface water concentrations and ammonia standards, concentrations of ammonia at a given location may exceed standards at one time and meet them at others.

Table 5–30. Temperature and pH-Dependent Values of the CCC (Chronic Criterion) for Fish Early Life Stages Present

CCC for Fish Early Life Stages Present, mg NH_3−N/L										
pH	Temperature C									
	0	14	16	18	20	22	24	26	28	30
6.5	6.67	6.67	6.06	5.33	4.68	4.12	3.62	3.18	2.80	2.46
6.6	6.57	6.57	5.97	5.25	4.61	4.05	3.56	3.13	2.75	2.42
6.7	6.44	6.44	5.86	5.15	4.52	3.98	3.50	3.07	2.70	2.37
6.8	6.29	6.29	5.72	5.03	4.42	3.89	3.42	3.00	2.64	2.32
6.9	6.12	6.12	5.56	4.89	4.30	3.78	3.32	2.92	2.57	2.25
7.0	5.91	5.91	5.37	4.72	4.15	3.65	3.21	2.82	2.48	2.18
7.1	5.67	5.67	5.15	4.53	3.98	3.50	3.08	2.70	2.38	2.09
7.2	5.39	5.39	4.90	4.31	3.78	3.33	2.92	2.57	2.26	1.99
7.3	5.08	5.08	4.61	4.06	3.57	3.13	2.76	2.42	2.13	1.87
7.4	4.73	4.73	4.30	3.78	3.32	2.92	2.57	2.26	1.98	1.74
7.5	4.36	4.36	3.97	3.49	3.06	2.69	2.37	2.08	1.83	1.61
7.6	3.98	3.98	3.61	3.18	2.79	2.45	2.16	1.90	1.67	1.47
7.7	3.58	3.58	3.25	2.86	2.51	2.21	1.94	1.71	1.50	1.32
7.8	3.18	3.18	2.89	2.54	2.23	1.96	1.73	1.52	1.33	1.17
7.9	2.80	2.80	2.54	2.24	1.96	1.73	1.52	1.33	1.17	1.03
8.0	2.43	2.43	2.21	1.94	1.71	1.50	1.32	1.16	1.02	0.897
8.1	2.10	2.10	1.91	1.68	1.47	1.29	1.14	1.00	0.879	0.773
8.2	1.79	1.79	1.63	1.43	1.26	1.11	0.973	0.855	0.752	0.661
8.3	1.52	1.52	1.39	1.22	1.07	0.941	0.827	0.727	0.639	0.562
8.4	1.29	1.29	1.17	1.03	0.906	0.796	0.700	0.615	0.541	0.475
8.5	1.09	1.09	0.990	0.870	0.762	0.672	0.591	0.520	0.457	0.401
8.6	0.920	0.920	0.836	0.735	0.646	0.568	0.499	0.439	0.386	0.339
8.7	0.778	0.778	0.707	0.622	0.547	0.480	0.422	0.371	0.326	0.287
8.8	0.661	0.661	0.601	0.528	0.464	0.408	0.359	0.315	0.277	0.244
8.9	0.565	0.565	0.513	0.451	0.397	0.349	0.306	0.269	0.237	0.208
9.0	0.486	0.486	0.442	0.389	0.342	0.300	0.264	0.232	0.204	0.179

Application of Ammonia Aquatic Criteria

Because the ammonia criteria change with water temperature and pH, and because criteria are established for both chronic and acute exposures, interpretation of ammonia data and application of aquatic criteria are somewhat complicated. Because of the variability in ammonia concentrations at monitoring locations both spatially and temporally, establishing where and when criteria should be met is not straightforward. In addition, both federal and state criteria are established for given averaging periods, and exemptions are provided for exceedences that occur during extremely low flow conditions.

Table 5–31.Temperature and pH-Dependent Values of the CCC (Chronic Criterion) for Fish Early Life Stages Absent

CCC for Fish Early Life Stages Absent, mg NH_3−N/L										
pH	Temperature C									
	0-7	8	9	10	11	12	13	14	15[a]	16[a]
6.5	10.8	10.1	9.51	8.92	8.36	7.84	7.35	6.89	6.46	6.06
6.6	10.7	9.99	9.37	8.79	8.24	7.72	7.24	6.79	6.36	5.97
6.7	10.5	9.81	9.20	8.62	8.08	7.58	7.11	6.66	6.25	5.86
6.8	10.2	9.58	8.98	8.42	7.90	7.40	6.94	6.51	6.10	5.72
6.9	9.93	9.31	8.73	8.19	7.68	7.20	6.75	6.33	5.93	5.56
7.0	9.60	9.00	8.43	7.91	7.41	6.95	6.52	6.11	5.73	5.37
7.1	9.20	8.63	8.09	7.58	7.11	6.67	6.25	5.86	5.49	5.15
7.2	8.75	8.20	7.69	7.21	6.76	6.34	5.94	5.57	5.22	4.90
7.3	8.24	7.73	7.25	6.79	6.37	5.97	5.60	5.25	4.92	4.61
7.4	7.69	7.21	6.76	6.33	5.94	5.57	5.22	4.89	4.59	4.30
7.5	7.09	6.64	6.23	5.84	5.48	5.13	4.81	4.51	4.23	3.97
7.6	6.46	6.05	5.67	5.32	4.99	4.68	4.38	4.11	3.85	3.61
7.7	5.81	5.45	5.11	4.76	4.49	4.21	3.95	3.70	3.47	3.25
7.8	5.17	4.84	4.54	4.26	3.99	3.74	3.51	3.29	3.09	2.89
7.9	4.54	4.26	3.99	3.74	3.51	3.29	3.09	2.89	2.71	2.54
8.0	3.95	3.70	3.47	3.26	3.05	2.86	2.68	2.52	2.36	2.21
8.1	3.41	3.19	2.99	2.81	2.63	2.47	2.31	2.17	2.03	1.91
8.2	2.91	2.73	2.56	2.40	2.25	2.11	1.98	1.85	1.74	1.63
8.3	2.47	2.32	2.18	2.04	1.91	1.79	1.68	1.58	1.48	1.39
8.4	2.09	1.96	1.84	1.73	1.62	1.52	1.42	1.33	1.25	1.19
8.5	1.77	1.66	1.55	1.46	1.37	1.28	1.20	1.13	1.06	0.990
8.6	1.49	1.40	1.31	1.23	1.15	1.08	1.01	0.951	0.892	0.836
8.7	1.26	1.18	1.11	1.04	0.976	0.915	0.858	0.805	0.754	0.707
8.8	1.07	1.01	0.944	0.885	0.829	0.778	0.729	0.684	0.641	0.601
8.9	0.917	0.860	0.806	0.756	0.709	0.664	0.623	0.584	0.548	0.513
9.0	0.790	0.740	0.694	0.651	0.610	0.572	0.536	0.503	0.471	0.442

[a] At 15 C and above, the criterion for fish early life stages absent is the same as the criterion for fish early life stages present.

The following discussion highlights some of the issues regarding application of ammonia surface water criteria and evaluating compliance with state surface water standards.

- State of Utah surface water regulations state that "the chronic mixing zone in rivers and streams shall not exceed 2,500 ft," which is defined from the most upstream point of contaminant discharge. For the Moab site, as identified by the USF&WS (1998), this upstream location for ammonia is approximately at the confluence of Moab Wash with the Colorado River.
- State of Utah surface water regulations state that concentrations in a mixing zone should not be "acutely lethal as determined by bioassay or other approved procedure." They further state that acute mixing zones "shall not exceed 50 percent of stream width nor have a residency time of greater than 15 minutes." It is assumed that, in the absence of site-specific testing, concentrations exceeding acute criteria equate to acutely lethal concentrations. It is further assumed that restrictions on acutely lethal concentrations mean that acute criteria must be met everywhere in and along a surface water body.
- Because of the more restrictive definition of the acute mixing zone, it is assumed that the acute criteria will be the driver for establishment of a ground water compliance strategy for the Moab site. It is assumed that if acute criteria can be met everywhere in the Colorado River, compliance with chronic criteria will be achievable at the downstream edge of the permitted chronic mixing zone.

Data for ammonia in surface water associated with the Moab site were analyzed to determine the scope of the contamination problem and help define ground water remediation goals, given the variability of observed ammonia concentrations and the variability of the applicable aquatic criteria. Ammonia surface water data collected between 2000 and 2002 were used in the analysis in this document. All ammonia analyses for which pH was also available were converted to total ammonia reported as N for ease in comparison to the federal criteria. Applicable federal chronic and acute criteria were then calculated for each sampling location and event using corresponding temperature and/or pH values to examine the variability of the standard that applies. Calculated values for criteria that apply are plotted in Figure 5–47 and Figure 5–48 for acute and chronic criteria, respectively. To normalize the data set for observed ammonia concentrations in order to compare spatially and temporally diverse data, the observed ammonia values for each data point were divided by the applicable chronic and acute criteria to determine how observations compared with criteria. Those data are shown in Figure 5–49 and Figure 5–50 for acute and chronic criteria, respectively. Data used in the analysis are included in Appendix D, Calculation X0047701.

Results of the ammonia surface water data show considerable variation, mostly because of the sample location and river flows that were present during the sampling events. Only those samples from locations adjacent to the site that have been cut off from the main flow of the river are significantly elevated in ammonia (see sample descriptions in Appendix C−1). Samples collected in the main river flow show no or minimal impact from ammonia resulting from discharge of contaminated ground water.

Preliminary statistics for applicable ammonia standards are presented in Appendix D, Calculation X0047701. These statistics assume data are normally distributed, which is not the case with this data set. However, the results are useful for interpreting preliminary ground water modeling results and for establishing target ground water remediation goals. Figure 5–51 is a histogram of acute criteria concentrations that apply at the Moab site for locations sampled since 2000. This figure and data included in Appendix D, Calculation X0047701 allow some generalizations to be made with respect to developing remediation goals for ammonia and evaluating results of the ground water modeling (Section 7.0).

With respect to the acute ammonia criteria, a criterion of less than 3 mg/L would apply to fewer than 20 percent of locations sampled (i.e., a concentration of greater than 3 mg/L applies to more than 80 percent of the locations). Figure 5–51 indicates that the 3 to 6 mg/L criteria range appears to apply in the majority of cases. Of the locations where a concentration of less than 3 mg/L was the applicable acute criterion, only 3 locations (approximately 1 percent) exceeded their respective criteria. Therefore, if a concentration of 3 mg/L ammonia-N could be achieved everywhere in surface water, 99 percent of the locations sampled in the past would be in compliance with applicable acute ammonia criteria. Figure 5–47, showing the distribution of applicable acute criteria concentrations, shows that in the backwater areas considered most important for endangered fish (described in Section 4.10.1, Figure 4−23), nearly all applicable criteria are above 3 mg/L. Therefore, if concentrations of ground water discharging to surface water can be reduced to the 3 to 6 mg/L range of total ammonia-N, it is highly likely that surface water compliance with both acute and chronic aquatic criteria can be achieved, considering effects of mixing with river water (see further discussion in Section 5.6.8) and the allowed chronic mixing zone.

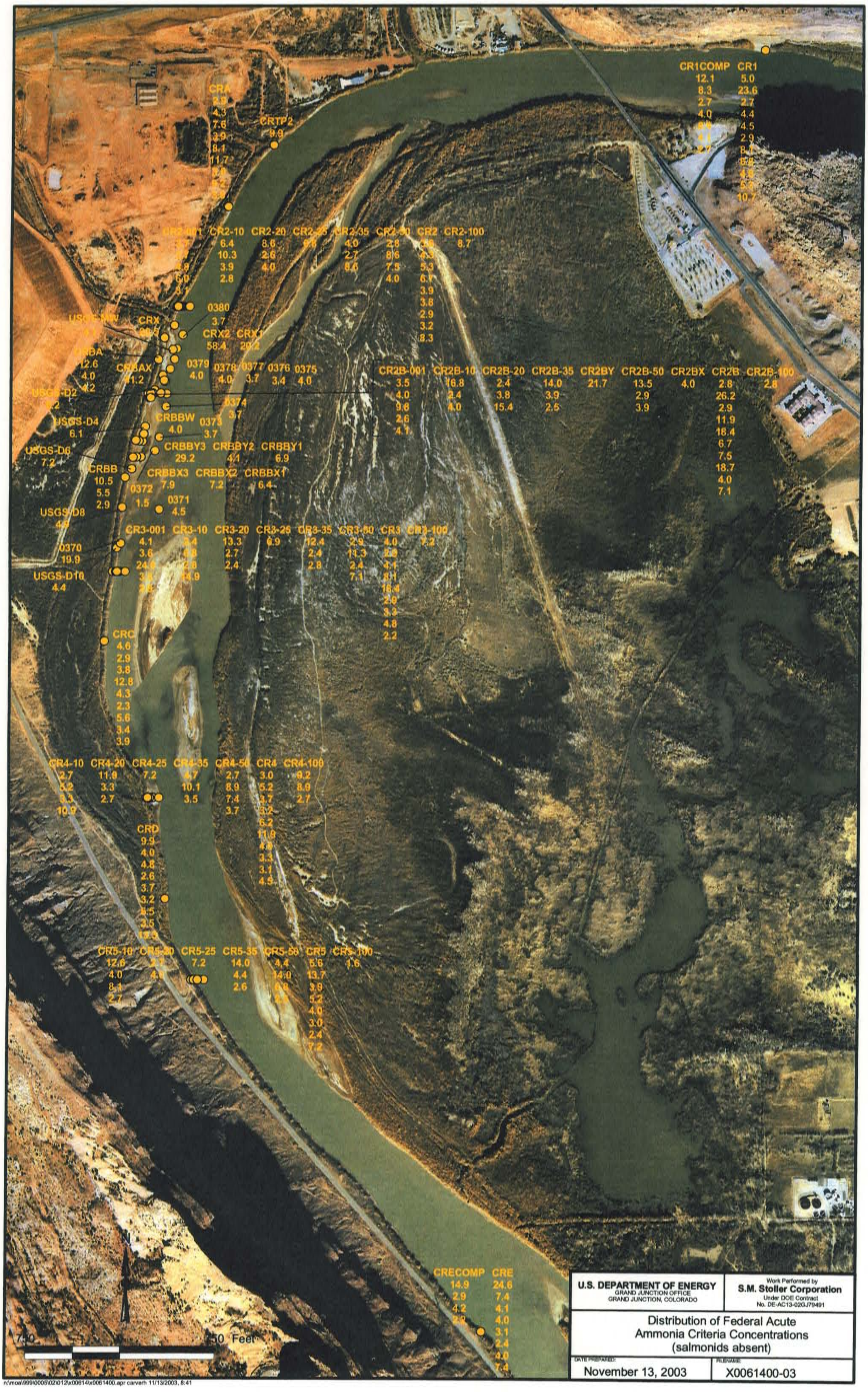

Figure 5–47. Distribution of Federal Acute Ammonia Criteria Concentrations (Salmonids Absent)

U.S. DEPARTMENT OF ENERGY
GRAND JUNCTION OFFICE
GRAND JUNCTION, COLORADO

Work Performed by
S.M. Stoller Corporation
Under DOE Contract
No. DE-AC13-02GJ79491

Distribution of Federal Chronic
Ammonia Criteria Concentrations
(young of year present)

DATE PREPARED:
November 13, 2003

FILENAME:
X0061400-04

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Figure 5–48. Distribution of Federal Chronic Ammonia Criteria Concentrations (Salmonids Present)

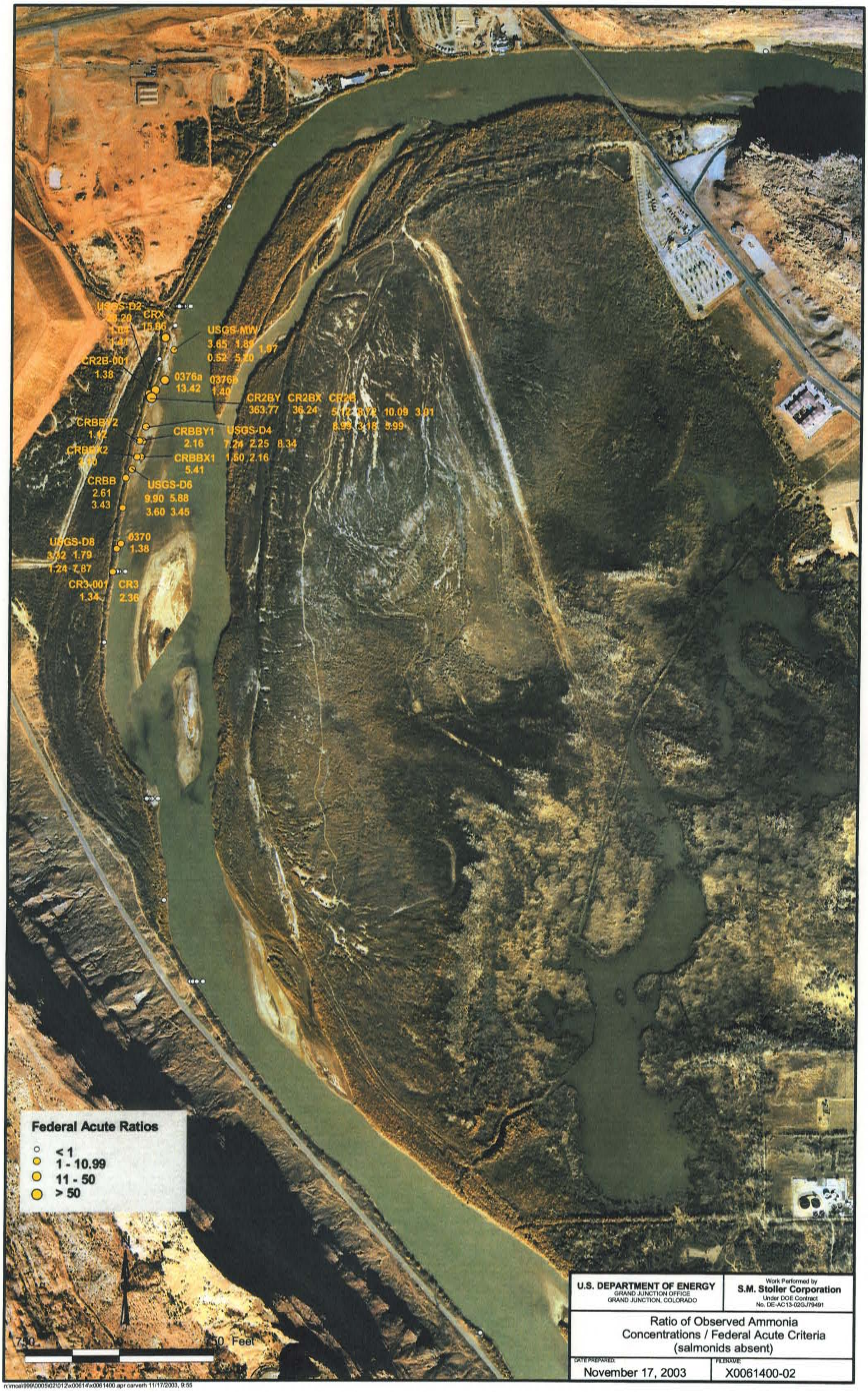

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Figure 5–49. Ratio of Observed Ammonia Concentrations/Federal Acute Criteria (Salmonids Absent)

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Figure 5–50. Ratio of Observed Ammonia Concentrations/Federal Chronic Criteria (Young of Year Present)

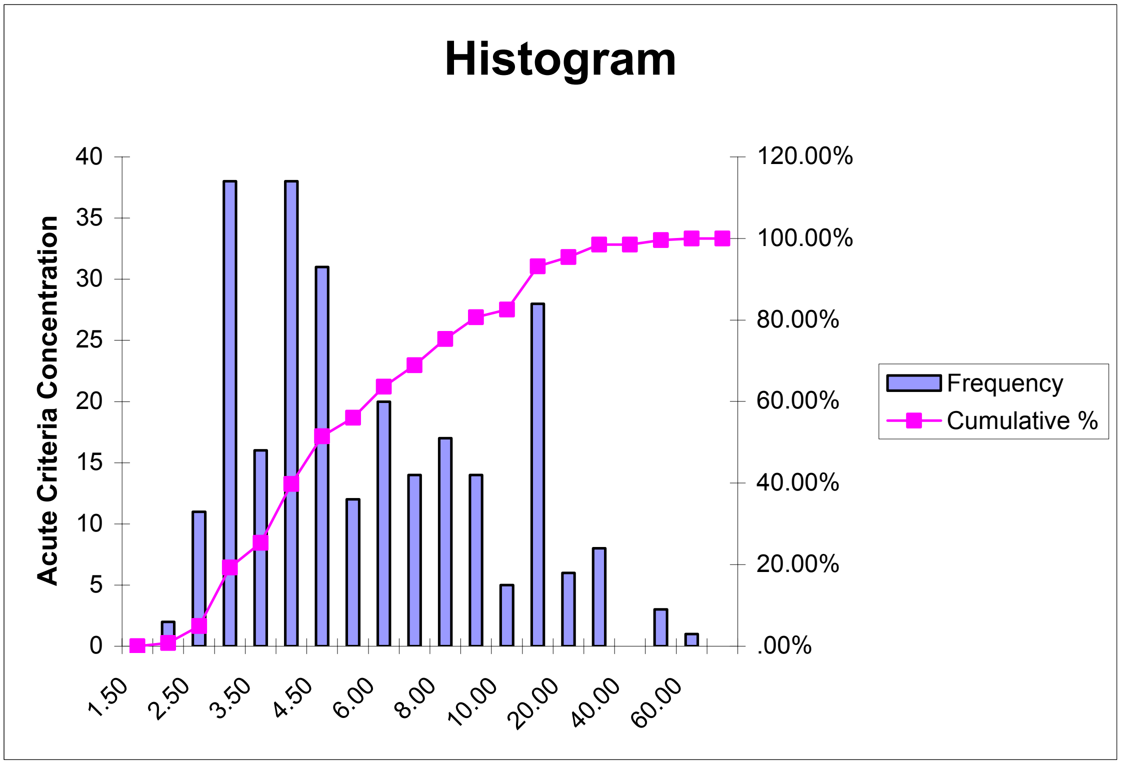

Figure 5–51. Histogram Showing Frequency Distribution of Federal Acute Ammonia Criteria and SMI Data

An evaluation of data presented in Appendix D, Calculation Number X0047701 indicates that a reasonable target range for chronic criteria is about 0.6 to 1.2 mg/L. Fewer than 20 percent of locations have chronic criteria below 0.6 mg/L; only about 30 percent of the calculated chronic criteria were above 1.2 mg/L.

Data sorted by decreasing values of the quotient of detected concentrations divided by acute criteria are presented in Appendix D, Calculation X0047701. Only a limited number of sampling locations had ammonia concentrations that exceeded the acute criterion for ammonia. All locations where concentrations exceed criteria are located immediately adjacent to the riverbank (see Figure 5–51). Most of the exceedences can be attributed to relatively few sample locations near the tailings pile. Those are mostly the same locations that were outside the range of background for other constituents, including major ions. This may reflect the localized influence of ground water discharge and/or the effects of evaporation of river water with isolated communication with the main portion of the river. Two locations downstream of the site—CRC and CR4—had concentrations of ammonia exceeding the chronic ammonia criteria on at least one sampling occasion. However, concentrations had decreased to levels below the standard at farther downstream locations, though ammonia was above detection in samples from location CRE.

Copper

Though SMI and DOE sampling did not identify copper as a major site-related constituent, ores were processed for copper recovery at the mill. Concentrations of 14 and 15 µg/L at two USGS sampling locations, D2 and D4, respectively, exceeded the Utah Water Quality Criteria for copper of 12 µg/L. These locations are in the slow-moving backwater area near location CR2B, where concentrations of numerous constituents are elevated. At most locations sampled in the past, copper has been at or below detection limits. Because copper could not be ruled out as a site-related constituent and because it was detected in samples collected by USGS, the Biological Assessment (DOE 2003b, Appendix A) retained copper as a contaminant of potential concern for the site. However, because of the limited data collected for copper, it is not discussed further.

Manganese

Figure 5–52 shows the distribution of manganese in surface water. Only those values exceeding a concentration of 0.01 mg/L (approximately the chronic aquatic benchmark) are labeled. Manganese levels exceed background adjacent to the site. The highest concentrations are detected immediately adjacent to the site in slow-moving or nonflowing waters just downstream of Moab Wash. Manganese was used in the milling process at the site and is known to be site-related. However, there is some evidence that significant levels of manganese occur naturally in ground water in the vicinity of the site. Some of the highest levels of manganese have been detected in shallow wells W1-4.3 and W1-7 in the Matheson Wetlands Preserve at concentrations of 38.5 and 19.1 mg/L, respectively.

Sulfate

Figure 5–53 shows the distribution of sulfate in surface water. Only values exceeding 500 mg/L are labeled. Highest concentrations of sulfate are in the same slow-moving waters or pools with elevated concentrations of other known site-related constituents. Somewhat surprisingly, given the high concentrations of sulfate in the ground water, relatively few locations have sulfate levels that exceed the upper limit of background range (439 mg/L). Most samples that have exceeded background have generally exceeded it only slightly.

Uranium

Figure 5–54 shows the distribution of uranium in surface water. Only concentrations exceeding 0.04 mg/L (the approximate acute aquatic benchmark) have been labeled. Although relatively few locations have concentrations that exceed this value, and the pattern is much the same as for other constituents, the high value at location CRA is notable in that it probably reflects the influence of the millsite uranium plume. However, the highest values are still from low-flow areas or stagnant pools (e.g., CR2BX and CR2BY). Concentrations exceeding background values are present from the area adjacent to the former millsite downstream to location CR5. Concentrations at downstream location CRE are in the range of background. Uranium concentrations above background are consistently detected at locations CR2, CR3, CR4, and CR5 and have been detected as far as 50 ft into the river channel at location CR3. Although the concentrations at those locations are probably not cause for concern, they do signal a site-related influence and are an indication that ground water discharge to surface water is affecting surface water quality.

5.6.6 Influence of Site-Related Ground Water on Surface Water Quality

As described in previously in this section, the Colorado River receives discharge from the ground water system except during periods of high flow. Discharge rate for the entire site was estimated at approximately 450 gpm (1.0 cfs) (Section 5.2.3). On the basis of this rate and even low river flow rates (e.g., 3,000−4,000 cfs), considerable dilution of site contaminants should occur as ground water mixes with surface water. This dilution is evident in historical data collected across transects adjacent to and downgradient of the site (SMI 2001, HLA 1998). Those studies showed that the influence of site ground water discharge on surface water quality was limited to areas near the bank of the river adjacent to the site and to areas immediately downstream of the site. In its final biological opinion, the USF&WS (1998) suggested that site-related contamination affected a 1.5 mile length of the river and encompassed approximately 7 acres of river surface area. A later study by SMI (2001) indicated a significantly smaller area of the river was affected (0.5 mile length, 3 acres of surface area).

There is no question that ground water discharge is affecting surface water quality. Known site constituents, most notably ammonia and uranium, are present in surface water adjacent to the site at concentrations much higher than those measured at background locations. Concentrations of sulfate and, to a lesser degree, manganese are also elevated above background levels. However, as discussed previously, manganese has been identified in high concentrations in brine ground water in the Matheson Wetlands Preserve and is not necessarily indicative of site-related contamination. Therefore, the remainder of this discussion will focus on ammonia, sulfate, and uranium as site-related constituents, with ammonia as the main indicator of site-related contamination.

Figure 5–55 through Figure 5–57 show results of shallow ground water sampling and analysis for ammonia, uranium, and sulfate in areas where surface water is most strongly affected by site-related contamination (inset on figures shows location of samples relative to the site. Samples were collected from shallow test pits at these locations, as described in Section 4.10.2. It can be seen that ammonia (total as N) is generally in the 300 to 500 mg/L range in the ground water. An approximate average concentration for uranium is about 2.5 mg/L, and sulfate is generally in the 7,000 to 10,000 mg/L range.

Figure 5–58, Figure 5–54, and Figure 5–53show maximum concentrations of these same constituents detected at surface locations since January 2000. A comparison of ground water data with surface water data shows that, with few exceptions, concentrations of site-related constituents are much lower in the surface water than in the ground water. Ammonia concentrations in the river appear to be approximately two orders of magnitude lower than in the ground water. Uranium and sulfate concentrations are generally 1 to 2 orders of magnitude lower in surface water.

Different mechanisms could cause attenuation of site-related constituents as they enter the surface water from the ground water system. Probably most significant is dilution due to mixing with river water. Ammonia could also volatilize or oxidize due to geochemical changes in moving from the ground water to surface water regime. Salt crusts observed along the riverbank indicate that some constituents are precipitating.

HLA (1998) estimated simplified dilution factors for ground water discharge based on their limited data set by comparing sample results from co-located ground water and surface water sampling locations. HLA ground water to surface water concentration ratios ranged from about 300 to 800, which are consistent with DOE's current observations.

USGS, in their ammonia toxicity study for the Moab site (Fairchild et al. 2002), collected co-located samples of soil pore water (i.e., ground water) and nearshore surface water from the Colorado River and analyzed them for selected constituents. These data were used to calculate dilution factors in a manner similar to HLA (1998) by taking the ratio of the pore water to the nearshore sample concentrations. Table 5–32 shows the calculated results for the ammonia data, calculated dilution factors range from less than 1 to over 1,000. The average dilution factor, excluding one very high outlier value, is 74. Though the USGS data may not have been collected for the purposes of determining dilution effects, that data set combined with observations regarding general ground water and surface water concentrations provide an indication that ammonia concentrations are attenuating as ground water discharges to the surface. Though the calculated dilution factors cover a considerable range, an order of magnitude dilution effect is considered reasonable and perhaps somewhat conservative compared to observed averages. Although isolated pools or very shallow areas may be exceptions, those locations are temporary and are unlikely to represent important aquatic habitat.

5.6.7 Matheson Wetlands Preserve/Brine Influence on Surface Water Quality

Ground water on the Matheson Wetlands Preserve side of the river, as well as the Moab site side, flows toward and discharges to the river. Data presented in this Section (5.6) indicate that discharge of site-related ground water has an effect on river water quality compared to background concentrations. Likewise, because of its difference from background river water quality (represented by location CR-1), water discharging from the Matheson Wetlands Preserve should also have some effect on river water quality. As noted in Section 5.3.4.4, quality of ground water associated with the Matheson Wetlands Preserve is highly influenced by the underlying Paradox Formation and is high in dissolved solids as well as numerous trace metals. Discharge of brine-influenced ground water may also be occurring on the Moab site side of the river, particularly in areas downstream of the tailings pile.

The Matheson Wetlands Preserve was the focus of a study conducted in 1993 (Cooper and Severn 1994) to evaluate the potential for enhancing the wetlands and using them as nursery areas for the endangered razorback sucker. Surface water and ground water hydrology data, water quality data, and biota data were all collected in support of this effort. The hydrologic studies indicated that distinct areas of the wetland could be delineated on the basis of the major hydrologic influences in the area. The studies found that adjacent to the river, the wetland and river hydrologic data were highly correlated, but that this correlation decreased with distance from the river. The Cooper and Severn wells M9, M10, and M11 represented locations where the river-ground water interaction was apparent. Water quality data for those locations were presented in Section 5.3.4.4.

Figure 5–52. Manganese Distribution in Surface Water
(Historic highs since 2000; labels are for locations exceeding 0.01 mg/L)

Figure 5–53. Sulfate Distribution in Surface Water
(Historic highs since 2000; labels are for locations exceeding 500 mg/L)

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Figure 5–54. Uranium Distribution in Surface Water
Historic highs since 2000; labels are for locations exceeding 0.04 mg/L

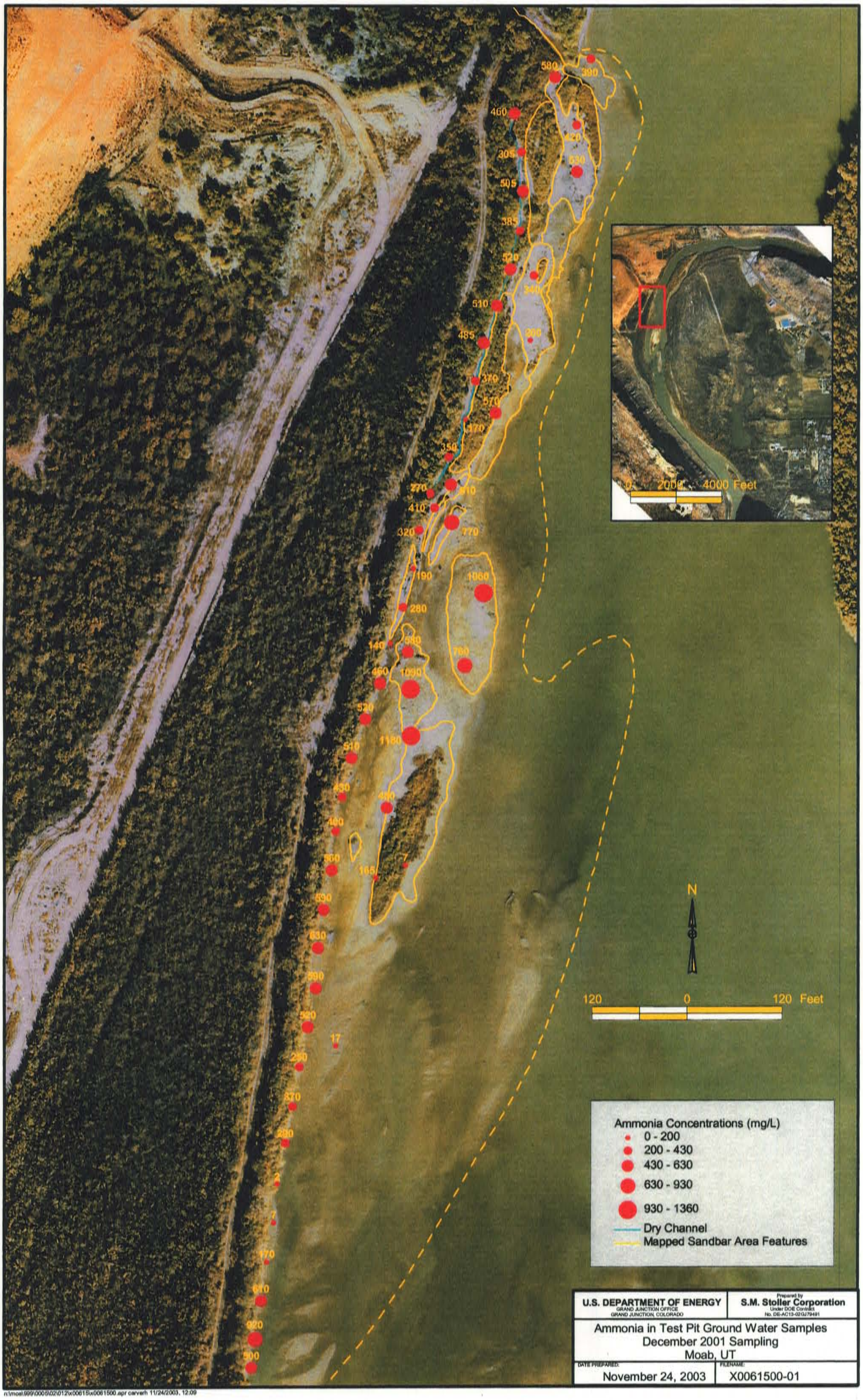


Figure 5–55. Ammonia in Test Pit Ground Water Samples, December 2001 Sampling

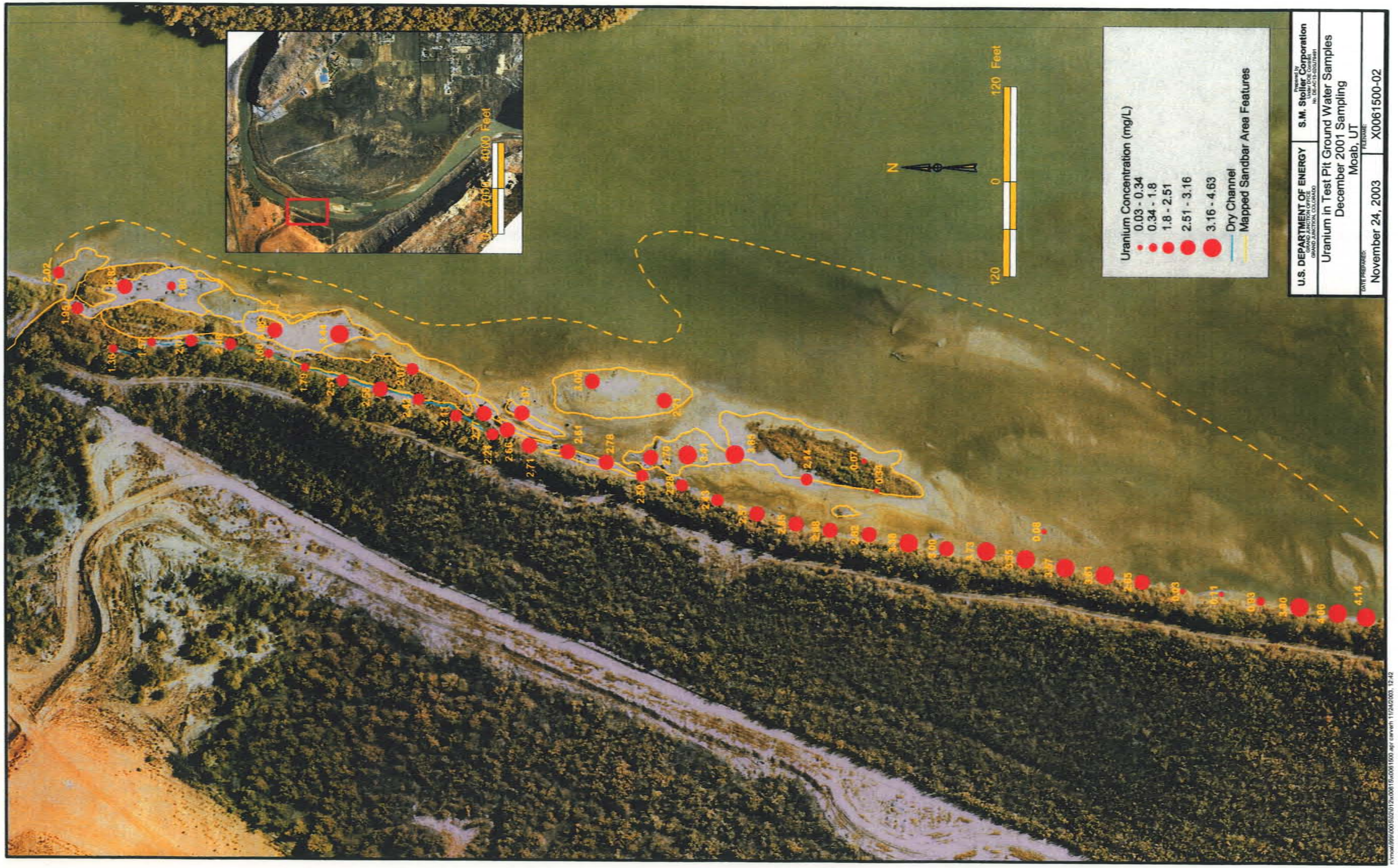


Figure 5–56. Uranium in Test Pit Ground Water Samples, December 2001 Sampling

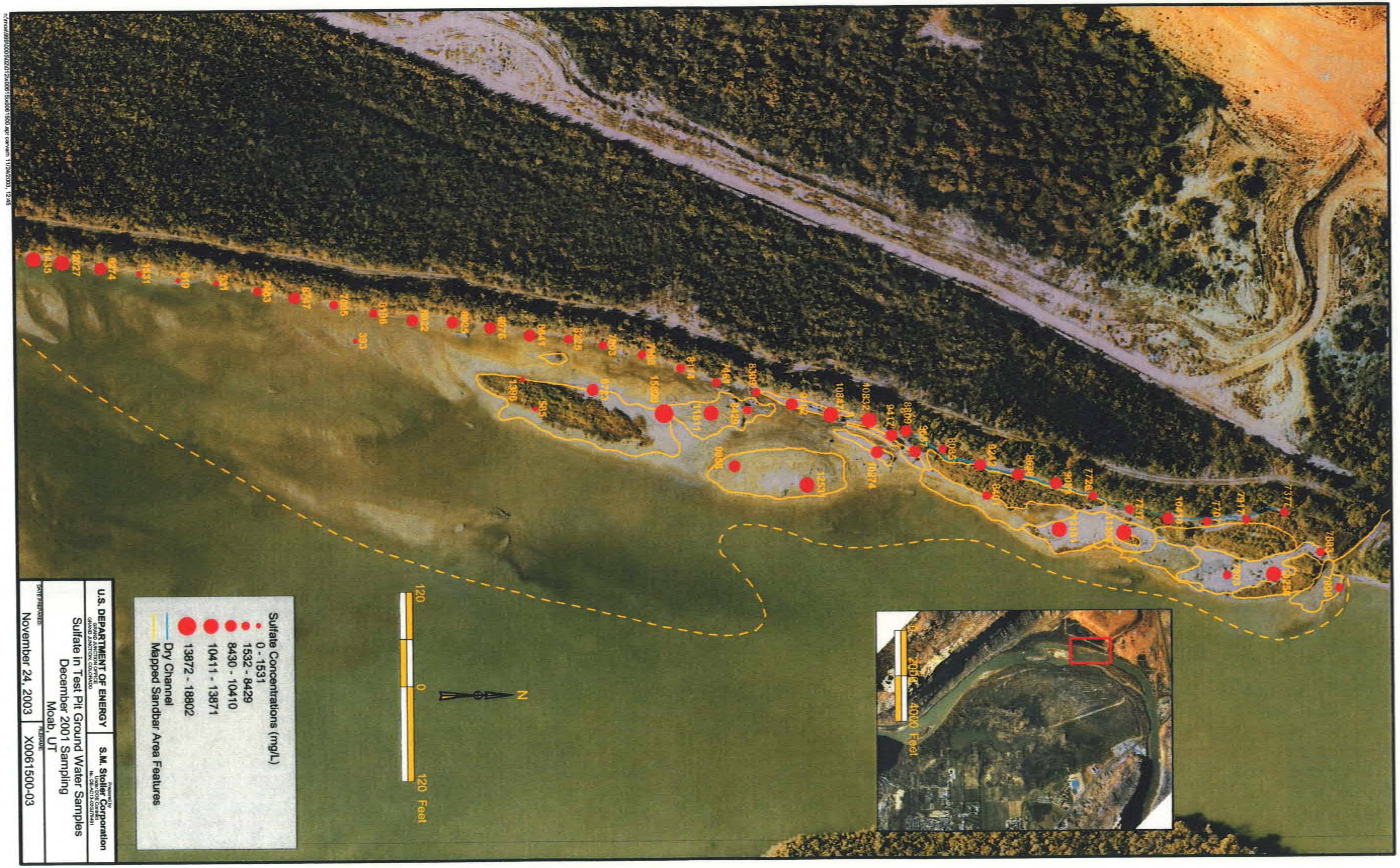


Figure 5–57. Sulfate in Test Pit Ground Water Samples, December 2001 Sampling

Figure 5–58. Ammonia Distribution in Surface Water
Historic highs since 2000; labels are for locations exceeding 0.6 mg/L

Table 5–32. Dilution Factors Calculated from Ratios of Pore Water to Nearshore Sample Concentrations

Location	Date	Soil Pore	Nearshore	Dilution Factor
U2	8/98	117	0.01	11,700[1]
U1	8/98	0	0	0
MW	8/98	477	20.5	23.3
D2	8/98	685	224	3.1
D4	8/98	771	34.9	22.1
D6	8/98	137	19.1	7.2
D8	8/98	0	4.59	0
D10	8/98	0	0.65	0
E4	2/99	1.98	2.22	0.9
E10	2/99	1.17	2.26	0.5
UX	2/99	2.25	2.17	1.0
U4	2/99	0	1.81	0
MW	2/99	492	6.57	74.9
D2	2/99	593	4.44	133.6
D4	2/99	499	9.26	53.9
D6	2/99	665	71.5	9.3
D8	2/99	43.7	35.7	1.2
D10	2/99	428	9.29	46.1
E4	9/99	0.11	0.21	0.5
E10	9/99	0.17	0.14	1.2
UX	9/99	0.14	0.21	0.7
U4	9/99	58.89	0.29	203.1
MW	9/99	652.67	14.95	43.7
D2	9/99	1082	3.81	284.0
D4	9/99	884.01	8.12	108.9
D6	9/99	587.31	0.43	1365.8
D8	9/99	13.35	0.36	37.1
D10	9/99	0.11	0.11	1
E4	2/00	1.3	0.21	6.19
E10	2/00	1.23	0.24	5.13
Hwy 191	2/00	0.91	0.84	1.08
UG	2/00	0.34	0.2	1.70
UX	2/00	63.6	0.85	74.82
U4	2/00	47.8	0.38	125.79
U2	2/00	177	1.19	148.74
MW	2/00	332	19.1	17.38
D2	2/00	602	2.02	298.02
D4	2/00	710	18.5	38.38
D6	2/00	705	41.4	17.03
D8	2/00	16.7	27.9	0.60
D10	2/00	303	5.18	58.49
D15	2/00	1.31	1.26	1.04
D20	2/00	0.41	1.17	0.35
E4	8/00	0.91	0.1	9.1
E10	8/00	2.25	0.12	18.75
Hwy 191	8/00	0.3	0.11	2.73
UX	8/00	0.39	0.08	4.88
U4	8/00	49.3	0.33	149.39
MW	8/00	519	1.39	373.38
D2	8/00	577	3.08	187.34
D4	8/00	617	39.1	15.78
D6	8/00	549	43.3	12.68
D8	8/00	6.66	9.06	0.74
D10	8/00	162	2.86	56.64
D15	8/00	1.01	0.7	1.44
D20	8/00	0.21	0.53	0.40
	mean	243.56	12.48	73.65

[1] Nearshore result was actually below detection limit

[2] Excludes first entry

After evaluating historical data, Cooper and Severn determined that in the past the influence of the river on the Matheson Wetlands Preserve was much stronger than it is currently. They noted that prior to construction of major dams along the Colorado River tributaries, river flows of greater than 40,000 cfs were common and would periodically flush out the wetland area, creating suitable conditions for larval and juvenile razorback suckers to thrive. However, since the late 1950s, flows of this magnitude are rare. As a consequence, the wetland water quality is more highly influenced by the naturally occurring ground water in the area, which is highly saline and influenced by the Paradox Formation. Wells adjacent to the river have elevated levels of trace metals as well as salts. Historical photographs showed that as the frequency of flooding of the wetlands diminished, the vegetation that once dominated the area—largely cottonwoods—began to die off and were replaced by more salt-tolerant species such as tamarisk, which dominate today.

Cooper and Severn concluded that the wetlands in the Matheson Wetlands Preserve could be used to enhance fish survival only if natural flood conditions could be restored or if water could be supplied to the area from a higher quality source and be protected from the brines in the ground water. Although the objective of this study was to evaluate the Matheson Wetlands Preserve as potential fish habitat, results do indicate that there is interaction of the ground water and Colorado River surface water, particularly in areas close to the river. The natural ground water that occurs in this area is highly saline and contains some metals in concentrations exceeding surface water standards. When this water is discharging to the river, it is likely to have some effect on river water quality, as is brine discharging from the Moab site side of the river. NRC (1999) noted that, on the basis of monitoring data from the State of Utah, TDS and manganese concentrations in the Colorado River increased downstream of the Moab site compared to upstream. However, they concluded that there could be many causes for these increases and provided calculations showing that manganese concentrations could not be attributed the tailings pile alone. Both TDS and manganese increases may represent the discharge of natural brines, which are elevated in both manganese and dissolved solids.

Very limited data have been collected in the river on the Matheson Wetlands Preserve side. Sampling done by the State of Utah in 1996 (see SMI 2001, Appendix A-1) focused on ammonia only and indicated that ammonia was not detectable in samples collected from the Matheson Wetlands Preserve side of the river. USGS sampled two locations on the Matheson Wetlands Preserve side during their studies (Fairchild et al. 2002). Ammonia was not detected in either of those samples, but there were detectable amounts of manganese, copper, and zinc. One sample collected by DOE in 2003 (location 215) had a chloride concentration elevated well above background (256 mg/L compared to a maximum background value of 172 mg/L), though other relevant constituents were within the range of background.

Although the data are not definitive, they provide good reason to conclude that discharge of Paradox Formation brine ground water to the Colorado River should have some effect on river water quality. Actual discharge rates and the degree to which water quality has been affected have not been measured. However, studies along the Dolores River where it crosses the Paradox Valley in Colorado show that the effect of brine discharge on that river is significant. Before brine withdrawal measures were implemented there, brine discharges were estimated to contribute about 313 tons/day of dissolved solids load (Chafin 2003).

5.6.8 Effects of Site-Related Contamination on Habitat

As described in Section 5.6.6, discharge of ground water with site-related contamination has had some effect on water quality in the Colorado River. Contaminant concentrations at some sampling locations have exceeded surface water standards or other aquatic benchmarks at certain times. Surface water quality is primarily of concern because of the potential for use of the area by endangered fish, particularly the Colorado pikeminnow, which is known to occur along that reach of the river. The Moab site is located within the 298-mile stretch of the Colorado River that has been designated as "critical habitat" for the Colorado pikeminnow. An additional 780 miles of the Green and the San Juan Rivers has also been designated critical pikeminnow habitat. In its 1998 biological opinion regarding proposed site reclamation (which was later withdrawn), the USF&WS concluded that the plan as proposed was likely to jeopardize continued existence of the species because it would continue to cause degraded water quality. Recovery plans for the pikeminnow and other endangered fish in the area cite the Moab tailings pile as a significant source of river degradation due to discharge of contaminated ground water. Ammonia, in particular, was mentioned as a major contaminant.

Data were collected and compiled as part of this SOWP to better understand the potential impact that ground water contamination associated with the site could have on areas critical to survival of the endangered fish. Potential exposure of pikeminnow to site-related contamination is directly related to the presence of suitable habitat and to the presence or absence of contamination in those potentially suitable areas. A number of studies have been conducted to better understand habitat use by the pikeminnow in several river systems, including the Colorado, Green, and San Juan rivers (Trammell and Christopherson 1999; Archer et al. 2000; Trammel and Chart 1998). It has been recognized that for the young of year pikeminnow, which are of most concern in the river adjacent to the Moab site, one of the important factors in contributing to suitable habitat is water velocity (DOE 2002g). Young of year pikeminnow are found primarily in low-velocity waters, which include backwaters as well as other habitats (Archer et al. 2000). However, despite the dependence of suitable habitat on river flow conditions, detailed studies of pikeminnow habitat in the Green River have shown that the quantity of habitat available does not vary significantly with the variation in river flow (Trammell and Christopherson 1999). Also, despite regulation of water flows at reservoirs and diversions of water for multiple purposes, total habitat area was not shown to have decreased since dams were completed in the mid-1960s (Trammell and Christopherson 1999).

Pikeminnow studies in the Colorado River have demonstrated that pikeminnow occupy a variety of habitats, from main portions of the river channel to slow-moving backwater areas (Trammell and Chart 1998). The preferred habitat can be characterized as secondary channels with wide mouths at the downstream end and an attached sand bar at the upstream end. Water enters the channel from downstream rather than flowing through the channel. Perhaps the most crucial habitat type for the species survival, however, is nursery habitat, which young of year pikeminnow rely on for their first year of life. Studies of nursery habitat characteristics have led to the conclusion that the major feature that defines essential nursery habitat is "persistence" (Trammell and Chart 1998). The most important pikeminnow habitats are relatively permanent and exist through fluctuations of river flow. These areas are considered crucial for over-winter survival of young of year. Although shallow habitats associated with migrating sand and sediment waves change continuously, the "quality" habitats can persist for several years (Trammell and Chart 1998).

On the basis of multiple studies of pikeminnow habitat, researchers have established a protocol for sampling backwater areas to monitor pikeminnow recovery efforts (Trammell and Christopherson 1999; Trammel and Chart 1999). The protocol calls for sampling backwaters with a minimum surface area of 30 m^2 (322 ft^2) and a minimum depth of 0.3 m (0.98 ft) for the Interagency Standardized Monitoring Program (USF&WS 1987). However, it has been noted that pikeminnow have been caught in areas as small as 19 m^2 (204 ft^2). The relatively permanent "average" secondary channel backwater areas noted above have mean surface areas of 999 m^2 (10,749 ft^2) and mean depths of 0.42 m (1.38 ft) (Trammell and Christopherson 1999).

Besides area and depth requirements, quality pikeminnow habitat must also be sufficiently turbid to provide adequate cover. Recent studies of pikeminnow in the Green River found a positive correlation of pikeminnow with higher turbidity; it was therefore recommended that a minimum depth for sampling in these turbid areas be reduced to 0.2 to 0.25 m (8−10 inches) (Day et al. 1999). A preference for temperatures somewhat warmer than the main river channel may also be important. However, in a study of the Colorado River pikeminnow nursery habitat, it was noted that fluctuations of temperature in backwater areas result in a lower mean daily temperature than in the main channel and that if pikeminnow closely follow temperature gradients, movement in and out of backwaters would be more frequent that previously assumed (Trammell and Chart 1998). Despite a concerted effort to better understand important pikeminnow habitat, it is not unusual for results of individual studies to contradict one another or be at odds with assumed pikeminnow preferences. Many of the habitat studies have been conducted in areas of the Green River. More limited studies along the Colorado River have produced some results that are inconsistent with Green River studies. Whether these results are due to differences in the river systems or simply unpredictability of pikeminnow behavior is not known.

Using the above descriptions of "quality" pikeminnow habitat and the characteristics of the Colorado River in the vicinity of the Moab site, the presence of suitable habitat can be determined. In 1995, as part of a multi-agency effort to develop a plan for studying the effects of the Moab site (then called the Atlas site) on endangered fish habitat, the river was surveyed upstream and downstream of the site to identify suitable habitat areas for comparison. Six good pikeminnow habitats were identified; three were upstream and three were downstream of the site. An area adjacent to the site was also proposed for sampling, even though the area was identified as "poor endangered species habitat"; sampling was proposed here because it was noted that "the fish need to be able to navigate up and down the shallow water habitat near the banks…" (DOI 1995). This area corresponds to that identified by Utah Department of Wildlife as potentially suitable (see Section 4.10 and Figure 4−23). Although this area can be suitable habitat at times, it does not meet the important criterion of persistence. In the 1995 survey, fair to very good habitat was identified in the river downstream of the Portal. Other possible habitat was identified on the opposite side of the river from the site; however, these areas were considered "less desirable" because of inadequate substrate or the fact that they were dry much of the time.

To date, no areas adjacent to the Moab site have been identified that correspond to the deep, scour-channel backwaters that are preferred by age-0 pikeminnow (Trammel and Christopherson 1999). This type of habitat is generally found below large sand bar complexes (Trammel and Christopherson 1999). The recovery plan for the pikeminnow (USF&WS 2002) indicates that much research attention has been focused on understanding backwaters and their importance to early life stage and age-0 fish. The plan describes these backwaters as follows:

> It is important to note that these backwaters are formed after cessation of spring runoff within the active channel and are not floodplain features. Colorado pikeminnow larvae occupy these in-channel backwaters soon after hatching. They tend to occur in backwaters that are large, warm, deep (average, about 0.3 m in the Green River), and turbid (...). Recent research (Day et al. 1999a, 1999b; Trammell and Chart 1999a, 1999b) has confirmed these preferences and suggested that a particular type of backwater is preferred by Colorado pikeminnow larvae and juveniles. Such backwaters are created when a secondary channel is cut off at the upper end, but remains connected to the river at the downstream end. These chute channels are deep and may persist even when discharge levels change dramatically. An optimal river-reach environment for growth and survival of early life stages of Colorado pikeminnow has warm, relatively stable backwaters, warm river channels, and abundant food (...)."

As previously mentioned, the sandbar located along the west bank of the Colorado River just south of Moab Wash (Figure 4-23, Section 4.10) extends approximately 1,300 ft downriver. A 2- to 4-ft-deep channel located between the riverbank and the sandbar was identified as having the greatest potential to develop into suitable Colorado pikeminnow habitat, depending on the river stage. During an average spring runoff in April and May, these areas (A, B, and C in Figure 4−23) become completely inundated. When the river recedes after the runoff, the sandbar becomes exposed, slow-moving backwaters form between the sandbar and the bank, and potentially suitable habitat can develop in the 2- to 4-ft-deep channel. As the river level further declines in the fall, the backwaters become isolated from the main channel and evaporate to dryness.

A cross-sectional view of the backwater channel that developed in Area A is shown in Figure 5–59 as an example. Plotted on the figure are the river stage when the channel becomes inundated and the river stage when the channel goes dry. These flows were determined by application of the regression equation presented in Section 4.4. It is apparent that potentially suitable habitat is not present in Area A when the river flow is greater than 12,500 cfs or less than 8,750 cfs. Habitat availability and quality depend upon the time of year, changes in river structure, and water level. Because these natural processes can physically alter the characteristics of river channel, the exact location of potentially suitable habitat can change seasonally or annually.

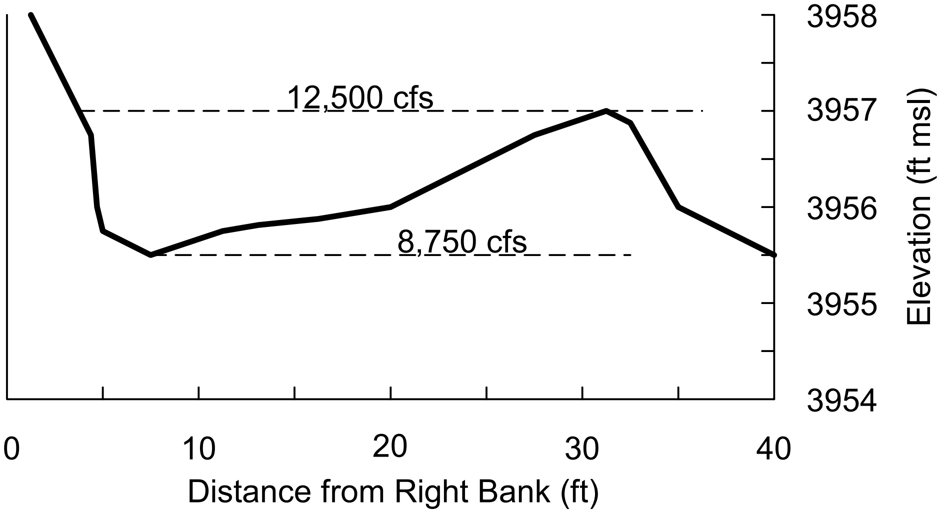


Figure 5–59. Cross-Sectional View of River Elevations in the Potentially Suitable Habitat Area A

Figure 5–60 depicts the number of post-peak days each year from 1959 to 2003 when river flows were between 8,750 and 12,500 cfs. This range of post-peak flows represents the stage during which potentially suitable habitat may develop in Area A. Also presented in Figure 5–60 is the peak flow for each year shown.

Historical data between 1959 and 2003 show that the number of yearly post-peak days with flows between 8,750 and 12,500 cfs ranges from 0 to 32 and averages 9 days. The majority of these post-spring-runoff peak days with flows providing potentially suitable habitat occur during June and July (82 percent) according to the historical data, and 12 percent of the days occurred in August. Only 5 percent of the days occurred in May, and 1 percent of the days where in September. The only year since 1959 in which flows were measured between 8,750 and 12,500 cfs in September was 1984, which had an extreme spring runoff peak of nearly 70,000 cfs.

Figure 5–60. Peak Flows and Corresponding Number of Days With Flows Between 8,750 and 12,500 cfs, 1959–2003

Figure 5–60 shows a wide range in the number of days during the year when flows range between 8,750 and 12,500 cfs. Although the seasonal peaking of flow is the main source of flows in this range, the duration of the peak is also heavily influenced by the weather systems of the Colorado River Basin during spring runoff. Cooler temperatures tend to increase the duration of the peak flows.

Data presented above indicate that the areas identified as most potentially suitable at the Moab site may actually have limited suitability. A comparison of river flow rates and periods of inundation of that location indicate that, on average, water is contained in those “backwaters” for only a 9-day period each year. These features are therefore floodplain features and not the important in-channel backwaters that are critical to the species survival. This supports the 1995 assessment that the area adjacent to the Moab site is "poor endangered species habitat," and that perhaps the main concern should be for fish that pass through that stretch of the river rather than use it for prolonged periods of time. If this is the case, the greatest concern should be with water quality in waters that are navigable to fish (or that would carry larvae downstream). Concentrations of concern would be those that could have an effect due to short term exposures—that is, acute concentrations. As described in Section 5.6.5, a reasonable range for acute ammonia criteria is approximately 3 to 6 mg/L.

Figure 5–58 shows maximum concentrations of ammonia detected in surface water since 2000. Appendix C, Section C.1 presents descriptions of some of those locations at the time of sample collection. It is notable that samples from most locations with a CR2 or CRB designation and location CR3, all of which have high concentrations of ammonia, were collected where water was only inches in depth, less than the Interagency Standardized Monitoring Program protocol criteria. It is possible that these samples were collected from portions of the river with characteristics similar to those shown in Figure 5–61, which shows an area between locations CR2 and CR2B. During most sampling events at most locations, maximum ammonia concentrations have generally been within the range of acute criteria, despite the fact that sampling conducted to date has been biased toward finding site-related contaminants. Ammonia concentration at all locations but one that were sampled more than once, have been below the acute and chronic criteria at one time. Only location CR3-001 has consistently had ammonia levels that exceeded chronic criteria.

Figure 5–61. Northernmost Mud Flat

5.6.9 Summary

The influence of site-related contamination appears to have been limited with respect to influence on surface water quality. Most of the samples with elevated concentrations of ammonia and other constituents do not represent river water samples in areas where aquatic organisms of concern would be expected. No habitat that would be considered “desirable” or “preferred” based on descriptions in the pikeminnow recovery plan (USF&WS 2002) has yet been identified in the site vicinity. It is possible that aquatic receptors could be exposed to elevated levels of site constituents very close to the riverbank, but concentrations drop off rapidly with distance from shore due to mixing with river water. Except for the cut off pools of water or areas that are only inches deep, concentrations of ammonia in surface water in the site vicinity are no higher than ammonia levels being discharged at the sewage treatment plant downstream and across the river from the site (concentrations at the outfall were 10 mg/L for a sample collected in 2003).

6.0 Site Conceptual Model

This section integrates the environmental characterization data that were described in previous sections and develops a site conceptual model for the project area and three potential end states for the site as evaluated in the EIS (DOE 2003b). Except for the management of the tailings pile, all of the elements of the site conceptual model are common to each proposed remediation alternative. A block diagram of the site that illustrates the major hydrologic components is presented in Figure 6−1. These components include areal recharge; lateral inflow from Moab Wash; inflow from the Glen Canyon aquifer, the Entrada Formation, and the Moab Fault; inflow from the tailings pile; outflow to evapotranspiration; and outflow to the Colorado River.

The three possible end states—no action, on-site disposal, and off-site disposal—differ in how the tailings pile is to be stabilized. The no-action alternative assumes that the drainage of pore water from the tailings pile will decline from its current rate of 20 gpm to a steady-state condition of approximately 8 gpm during the next 20 years. No additional engineering measures would be taken to stabilize the tailings pile. Existing conditions for the tailings pile without any future changes are portrayed in Figure 6−1.

The on-site and off-site disposal alternatives assume that hydrologic processes for the tailings pile would change. Details of how these alternatives would affect the infiltration of water through the tailings pile, and inflow of chemical mass, are described in Section 6.8.

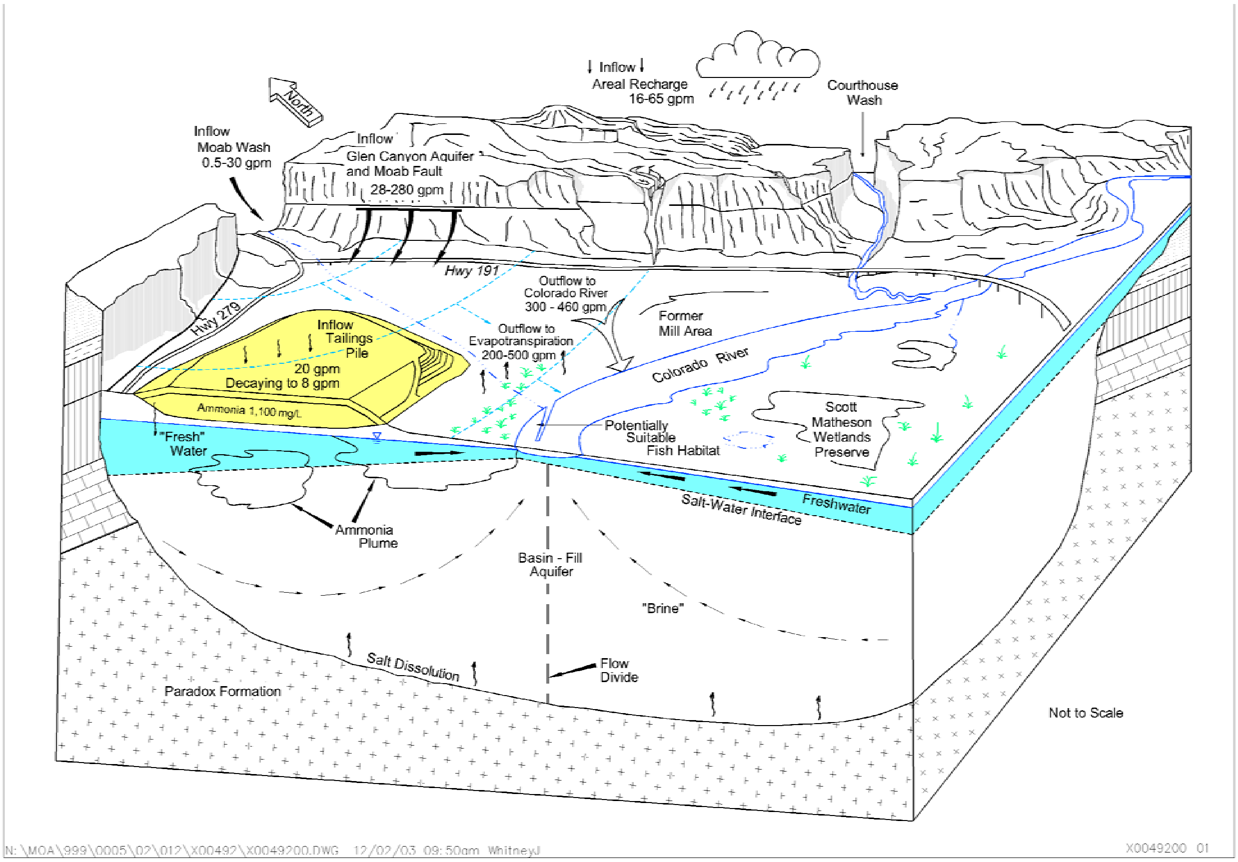


Figure 6–1. Conceptual Model—Existing Site Conditions and No-Action Alternative

6.1 Ground Water Hydrology

Ground water in the Moab region occurs in the unconsolidated Quaternary material deposited on the floor of Moab and Spanish Valleys and in consolidated bedrock formations. Unconsolidated alluvial deposits overlie mostly the Paradox Formation at the site and comprise two distinct depositional facies: the Moab Wash alluvium and the basin-fill alluvium. The Moab Wash alluvium includes fine-grained sand, gravelly sand, and detrital material that has traveled down Moab Wash and grades and interfingers, near the northwest boundary of the site, into the basin-fill alluvium that was deposited by the ancestral Colorado River.

The basin-fill alluvium is subdivided into two units; an upper unit and a lower unit. The upper unit is composed of fine sand, silt, and clay that ranges in thickness from 15 ft near the Colorado River to approximately 40 ft near the northern and northwestern boundary of the site. The upper unit is referred to as the silty-sand unit and is mostly Colorado River overbank deposits. The hydraulic conductivity for the silty-sand unit is typically less than 2 ft/day. The lower basin-fill alluvium consists mostly of gravelly sand and sandy gravel with minor amounts of silt and clay. This lower alluvial unit, referred to as the gravelly unit, thins and pinches out to the northwest along the subsurface bedrock contact and thickens to the southeast to over 450 ft thick near the deepest part of the basin. Average hydraulic conductivity of the gravelly unit of the basin-fill alluvium appears to range from 100 to 150 ft/day.

Much of the freshwater recharge entering the basin-fill aquifer comes from bedrock aquifers that bound the basin-fill aquifer along its northern margin. The fresh water comes from the Entrada aquifer and the Glen Canyon aquifer. Assuming a vertical hydraulic conductivity of 0.5 ft/day and a vertical hydraulic gradient of $-3.6 \text{ x}10^{-3}$, the upward component of specific discharge is approximately 1.8×10^{-3} ft/day. However, depending on the local characteristics, such as degree of fracturing or proximity of faulting, the specific discharge from bedrock to alluvium along the northern margin of the site could be higher.

The Chinle, Moenkopi, and Paradox Formations form aquitards below the Glen Canyon aquifer. The Paradox Formation subcrops below alluvium at most of the site except for a small area along its northern and western boundaries, where Chinle Formation, Moenkopi Formation, Cutler Formation, and the Honaker Trail Formation are in direct contact with the alluvial aquifer.

The basin-fill aquifer contains natural brine (salt water) that underlies a relatively thin section of brackish to fresh water. Dissolution of the salt within the Paradox Formation is mostly responsible for the brine within the basin-fill alluvium. The interface that separates the upper fresh and brackish water from the brine is assumed to exist where the TDS concentration equals 35,000 mg/L. The position of the saltwater interface shifts vertically and laterally in response to hydrologic stresses such as evapotranspiration from tamarisk plant communities and changes in the stage of the Colorado River.

It is apparent from ground water level contours presented in Figure 5−13 that the direction of shallow ground water flow at the site is southeast. The shapes of the contours representing water levels of 3959 and 3961 ft amsl indicate possible local inflows of fresh water from bedrock aquifers to the alluvium along the northern boundary of the aquifer. Deflection of the 3,955 ft elevation contour near Courthouse Wash indicates another local inflow of water to the alluvium in this area through either the Wingate Sandstone or the Chinle and Moenkopi Formations, all of which subcrop below the alluvial aquifer in this area.

6.2 Water Budget

The spatial distribution of various water-budget components at the site is illustrated in Figure 6−1. Fresh water enters the site from the Glen Canyon and Entrada aquifers and presumably along the Moab Fault. Less significant sources of recharge to the site are infiltration of precipitation and runoff in addition to lateral inflow from Moab Wash. A relatively small quantity of contaminated water enters the alluvial aquifer as a result of seepage of tailings pore fluids from the base of the tailings pile. The Colorado River occasionally loses water to the alluvial aquifer in the form of temporary bank storage, such as during periods of high runoff. The magnitude of the inflow from the Colorado River to the alluvium is unknown.

Total inflow to the basin-fill aquifer at the Moab site is estimated to be between 65 and 400 gpm. Areal recharge is estimated to range from 5 to 20 percent of average annual precipitation and accounts for approximately 16 to 65 gpm of recharge to the aquifer. Lateral inflow from Moab Wash to the basin fill is estimated to range from 0.5 to 33 gpm. Inflow to the basin-fill alluvium by way of the Glen Canyon and Entrada aquifers is estimated to range from 28 to 280 gpm. Under current conditions the seepage of pore fluids from the tailings pile is approximately 20 gpm.

Outflow from the site occurs through evapotranspiration from the tamarisk grove located between the tailings pile and the Colorado River. The ET component of discharge accounts for approximately 200 to 500 gpm of outflow from the basin-fill aquifer. Outflow also occurs through discharge to the Colorado River. This Colorado River-discharge component is probably in the range of 300 to 460 gpm. The sum of these two outflow components comprises approximately 500 to 960 gpm. Collectively, the inflow and outflow components cover a wide range of values. The disparity between maximum estimated inflow (400 gpm) and minimum estimated outflow (500 gpm) suggests that the actual water budget might lie somewhere between the two reported ranges, or 450 gpm.

6.3 Chemical Transport Processes

The dominant dissolved ammonia species in the ground water contamination plume is ammonium ion (NH_4^+). Due to the high sulfate concentration, significant ammonium sulfate ($NH_4SO_4^-$) is also present. Under oxidizing conditions, ammonia reacts to form nitrite (NO_2^-), nitrate (NO_3^-), or nitrogen gas (N_2). Some of the transformation reactions are catalyzed by microbiological activity. Ammonia (mainly) and nitrate were used during the milling process at the Moab site. Ammonia and other constituents probably became concentrated as the tailings pile pond fluid evaporated to dryness when the mill went on standby and processing operations ceased. Under these conditions, an ammonia-bearing salt layer precipitated in the upper 10 ft of the tailings pile.

Ammonia is a strong cation exchanger on clay minerals that are present in most aquifers. At pH values close to 9 dissolved ammonia is dominated by the uncharged ammonia species (NH_3^0) and is volatile. Ammonia is also a plant nutrient and is being used by tamarisks in the floodplain area.

As contaminated ground water migrates downgradient, the contaminants are subject to dispersion and transfer between the solid and liquid phases. These geochemical phenomena cause contaminants to travel at a slower rate than the average ground water velocity. Consequently, the time required for contaminants in the alluvial system to flush naturally out of the aquifer can be greater than the time required for passage of one pore volume of ground water.

This section presents a discussion of the bulk transport parameters, hydrodynamic dispersion, distribution coefficient (K_d), first-order decay, and porosity, which are often used to predict the fate and transport of ground water constituents.

6.3.1 Hydrodynamic Dispersion

Hydrodynamic dispersion is a parameter with two components that describes the longitudinal and transverse spreading that occurs when solutes are transported by the bulk motion of flowing ground water. The mechanical component is the product of dispersivity, which is a transport-model input parameter, and average linear velocity. The other component, diffusion, occurs because of thermal-kinetic energy of the solute particles; it is important at low velocities (Freeze and Cherry 1979). The diffusion component at the Moab site is important below the saltwater interface, where ground water flow velocities are very low. The sum of the mechanical and diffusive components equals the coefficient of hydrodynamic dispersion.

The literature on dispersivity as it relates to large-scale models is vague and often contradictory, with longitudinal values ranging from 2 percent to 30 percent of the length of the plume or maximum flow direction. Dispersivity is almost impossible to measure in the field for large sites. For the Moab site, values of 25 meters for longitudinal dispersivity and 2.5 meters for transverse dispersivity are assumed. These values relate to a longitudinal dispersivity of 5 percent of the model scale (maximum flow direction) and a transverse dispersivity of 10 percent of longitudinal dispersivity and are consistent with dispersivity reported for other large-scale transport models (Gelhar et al. 1992).

6.3.2 Distribution Coefficient

The distribution coefficient (K_d) is a bulk parameter that describes the retardation of contamination in aquifer systems. Most numerical ground water models use K_d in simulations of contaminant transport. The K_d is approximated from the empirical distribution ratio (R_d). The R_d for the Moab site was determined from laboratory testing described in Section 4.7. It is assumed that K_d is equivalent to R_d. The assumption is valid if (1) the system is always in chemical equilibrium, (2) if adsorption is the only chemical mechanism, and (3) if the R_d (or K_d) is independent of solution composition.

Laboratory measurements to determine the R_d for selected analytes were performed on alluvial and bedrock material to characterize subsurface contaminant transport processes at the Moab site. R_d values for ammonium (NH_4) are mostly between 0.5 and 3 mL/g at the Moab site. A literature search was conducted to obtain other K_d values for ammonia. All literature values for K_d were less than 1 (Ceazan et al. 1989; Kipp 1987), indicating that the low end of the site-specific range determined for Moab might be applicable. A value of 0.5 mL/g was assigned to the Moab site which is consistent with site-specific results and values obtained through the literature search.

6.3.3 First-order Decay

Oxidizing conditions in combination with microbiological activity are believed to exist beneath the tailings pile and within the aquifer in general. Under these conditions, ammonia species react to form nitrite, nitrate, or nitrogen gas. Literature values (Kipp 1987) ranging from 1.8×10^{-4} to 3.6×10^{-4} day^{-1} were obtained to describe the transformation of ammonia to its oxidation products at the Moab site. Because the range of values used by Kipp (1987) were derived from site conditions unique to another area, they would probably be different for the geochemical conditions at the Moab site. A first order decay value for ammonia of 1.8×10^{-4} day^{-1} was assumed for the Moab site.

6.3.4 Porosity

Porosity is a required parameter in modeling simulations because it enters into transport calculations not only in the seepage velocity term, but also in expressions for the solute mass in a given volume of aquifer and the rate at which that mass changes with time (Zheng and Bennet 1995). Porosity was not determined specifically for the Moab site. Rather, literature values compiled by Morris and Johnson (1967) were used because their values are regarded as reputable averages that span a wide variety of lithologic materials are widely used in the field of hydrology. On the basis of these published values, bedrock materials were assigned a porosity of 20 percent, and alluvial materials were assigned a porosity of 30 percent.

6.4 Ammonia Plume

The ammonia plume is partitioned with approximately 60 percent of the total mass of ammonia within the domain of the brine and 40 percent above it. The ammonia plume beneath the saltwater interface is believed to have formed when tailings pond fluids with very high TDS migrated downward on account of driving head and high density and equilibrated with the surrounding brine. Density driven flow explains the high concentrations of ammonia found 200 ft below the base of the tailings and the trend of increasing concentrations versus depth that are observed below both the tailings pile and the tamarisk grove. The ammonia plume above the saltwater interface formed as pore fluids draining from the tailings pile, and the surrounding toe drains, passed through the fresh/brackish-water zone of the basin–fill alluvium, leaving dissolved remnants of the ammonia distributed vertically throughout the fresh/brackish zone.

The ammonia plume encompasses broad reaches of the site both horizontally and vertically. The eastern terminus of the plume appears to coincide with the western bank of the Colorado River, perhaps because data from the aquifer beneath the river is limited; however, because the brine interface intersects the riverbed near the west bank of the river, the fresh/brackish water aquifer, and the eastward extension of the ammonia plume beneath the river, are essentially cut off by the underlying brine.

Figure 5−33 shows how the ammonia plume is distributed in the shallow ground water. The highest concentrations in the shallow ground water, which range from 500 mg/L to over 1,000 mg/L, extend from the eastern toe of the tailings pile to the Colorado River. In this area, the ammonia is most likely associated with seepage from the former toe drains. Ammonia is virtually absent beneath the tailings pile in the shallow ground water because first-order decay is assumed to have consumed it and through dilution by fresh water entering the site upgradient.

Figure 5−34 shows the ammonia concentration interpolated over a layer 50 ft beneath the equipotential surface. The plot shows that ammonia concentrations increase with depth at the site, particularly near the Colorado River, where the 1,000 and 2,500 mg/L isopleths are located. These site-wide maximum ammonia concentrations occur near the saltwater interface. In addition, a zone of elevated ammonia concentration also exists northeast of the pile beneath the area that was formerly occupied by the mill.

Figure 5−35 shows the ammonia concentrations that were interpolated at a depth of 150 ft beneath the equipotential surface. In the eastern half of the site this zone represents a region significantly below the saltwater interface. Beneath the tailings pile, the saltwater interface cuts through the ammonia plume at the 150-ft depth, and ammonia concentrations range from at least 50 mg/L to over 500 mg/L. Ammonia concentrations also exceed 1,000 mg/L at the 150-ft depth near the Colorado River.

A portion of the ammonia plume above the saltwater interface is probably metabolized by the tamarisk grove west of the riverbank, and the remaining ammonia discharges to the Colorado River. Ammonia contamination that resides below the saltwater interface is believed to discharge, by way of the riverbed, to the Colorado River. The midpoint of the river forms a line of symmetry where the natural flow components on the west site of the river are being balanced by mirror-image flow conditions on the east side. With ground water flowing toward the river from both sides, the river behaves as a line sink both for fluid and chemical mass. On the basis of this conceptual model, there is no exchange or movement of fresh/brackish ground water beneath the river, and the eastward limit to chemical-mass migration in the brine zone is one-half of the distance beneath the river.

6.5 Tailings Pile Evolution

A conceptual model for the evolution of tailings chemistry is proposed as follows:

1. During the acid milling operation from 1956 to 1961, milling fluids (TDS about 24,600 mg/L) were continuously discharged to the tailings pile and continuously buried by slurried tailings without significant evapo-concentration (NRC 1999). Thus, the lower portion of the tailings contains pore water with geometric-mean concentrations of: 24,619 mg/L TDS, 1,100 mg/L NH_3-N, and 7.87 mg/L U (Appendix D, Calculation X0025700). These values are consistent with pore water concentrations measured at the Monticello, Utah, site. The acidic mill effluent was neutralized with lime (perhaps incompletely at times), and the final pH values were variable, ranging from 4 to 10. River water was being used in the mill at about 1,300 gpm, and much of the mill effluent was discharged back to the Colorado River.
2. From 1961 to 1974, alkaline milling was used (for a brief period both alkaline and acid milling were used). The plant was using approximately 1,228 gpm of water, and approximately 188 gpm was still being discharged to the river (Table 3−4). Tailings pore water was similar in composition to that in Item 1 except that the pH was higher (due to the alkaline milling), ranging from about 8 to 10.
3. From 1974 to 1981, acid and alkaline milling were used. Water use was reduced to 130 gpm because water recycling was initiated (Section 3.5.1.2). Tailings pond evaporation kept up with mill effluent, and no water was discharged to the river. Lower water use resulted in

increased residence time in the tailings pond, leading to increased evaporation and higher concentrations of solutes in the tailings pore fluids.

4. From 1981 to 1984, only acid milling was used. This period shows up as lower paste pH measurements in the upper 10 ft of tailings (SRK 2000; Appendix D, Calculation X0025700). Low water usage and high evaporation continued to cause high concentrations of solutes in the pore fluids.
5. The mill closed in 1984. From 1984 to 1990, mill effluent remaining in the tailings pile pond underwent extensive evaporation. In 1990, pore water was withdrawn from the tailings through wells and evaporated at the surface, mostly in the slimes facies area. The pond was evaporated dry by 1993 (Pattison 1996). Evaporation of pore fluid expressed through tailings consolidation continued until 3 to 6 inches of salt crust formed over the slimes by summer 1995 (Pattison 1995). The crusts provided a base for the interim cover placed in 1995–1996. These activities caused high concentrations of solutes and deposition of salts in the upper portion of the tailings. Pore water in the uppermost layer is much higher in salt content than the lower layer. Geometric mean concentrations of TDS, NH_3-N, and U in pore fluid from samples believed to be at or near equilibrium with salt crusts are 213,748, 18,106 and 4.89 mg/L, respectively. The concentrations vary spatially and appear to be lower in the outer portion (sand facies) of the tailings.
6. The tailings pore water currently migrating from the bottom of the tailings pile has a composition of about 24,600 mg/L TDS, 1,100 mg/L NH_3-N, and 7.87 mg/L U. In the future, as water infiltrates the upper portion of the tailings, it will dissolve salt deposits, and pore water concentrations could be similar to those measured in the uppermost lysimeter samples and the sump (about 213,748 mg/L TDS, 18,106 mg/L NH_3-N, and 4.89 mg/L U). These high concentrations would persist as long as salt deposits remain in the tailings. After the salt deposits become depleted by dissolution in infiltrating water, pore water concentrations would decrease dramatically. Because of the dish-shaped geometry of the tailings pond, the salt deposits probably exist mostly in the central (slimes facies) portion of the tailings pile.
7. The chemistry of the acidic pore water will likely change as it percolates down through alkaline tailings. Solution pH will increase, and some minerals will form due to reaction with minerals such as calcium carbonate.

On the basis of the conceptual model provided above, the tailings pore water is thought to have at least two average compositions: higher TDS concentrations in the upper 10 ft of the slimes and lower concentrations for the remainder of the pile. Pore water samples with the highest TDS values are assumed to represent pore water that is at (or nearly at) equilibrium with the soluble salts that form thick mineral crusts in the upper 10 ft of the slimes facies. Samples selected to represent this unit are 0539, 0540, 0541, and the two sump samples; TDS concentrations in these samples range from 152,000 to 240,000 mg/L (Appendix D, Calculation X0025700).

All other pore water samples are used to represent the other units. In contrast to TDS and ammonia, uranium concentrations are not much higher in the sump sample than in samples of pore water from the lower portion of the tailings. Uranium concentrations in the pore water may be controlled by mineral saturation with oxidized uranium minerals. As acidic pore water with high ammonia levels moves down through high-pH, carbonate-bearing tailings, chemical precipitation will likely occur and concentrations of some constituents will decrease. Geometric means of the sample concentrations are used as the basis for the tailings pore-fluid concentrations provided in Table 6−1.

Table 6–1. Tailings Pore-Fluid Concentrations

	Upper 10 ft of Slime Facies	All Other Units
NH_3-N (mg/L)	18,100	1,100
U (mg/L)	4.89	7.87
TDS (mg/L)	214,000	24,600

6.6 Ground Water and Surface Water Interactions

The primary surface water body of concern for the Moab site is the Colorado River, which forms the site’s eastern boundary. As discussed in Section 5.0, the interaction of ground water and surface water produces unique surface water chemistries along the bank of the Colorado River and can influence the quality of habitat for the Colorado pikeminnow and other aquatic organisms that inhabit the river. The Colorado River combines a surface water component that flows past the Moab site with a ground water discharge component that empties into and mixes with the river water. The resulting ammonia concentration of the surface water may be thought of as a blend of two flow components: (1) a large quantity of river water at background ammonia concentration, and (2) a small quantity of ground water with elevated ammonia concentrations, that can be 1,000 mg/L or higher. The resulting ammonia concentration near the bank of the river could be relatively low if the measurement is taken at a time with a high flow rate in the river, or relatively high if the measurement is taken at a time with a low flow rate in the river.

Computed pore water velocities from a 2D cross-sectional model indicate that nearly all of the ground water that enters the river does so along a relatively narrow portion of the riverbed that coincides with the west bank. Moreover, the ground water discharge gradually decreases with distance toward the center of the river. Because the highest mass of site-related ammonia enters the Colorado River at the location that the minimum quantity of the surface water exists, there is a tendency for the ammonia concentrations in the surface water to exceed regulatory limits, especially along the west bank of the river. In addition to the mixing of ground water and surface water, the resulting concentrations can increase further in cutoff channels, low- or slow-moving (backwaters) shallow areas, and stagnant pools. This is because the diluting effect of additional surface water is minimized in these areas, and the concentrating effects of evaporation and continued ground water discharge are maximized.

Concentrations of ammonia in surface water vary considerably, mostly because of the sampling location and river flows that exist during sampling events. Only those samples from locations adjacent to the Moab site, which have been cut off from the main flow of the river, are significantly elevated in ammonia. Samples collected in the main river channel show no or minimal impact from ammonia resulting from discharge of site-contaminated ground water. On the basis of these relationships, it is projected that if the ammonia concentration in ground water discharging to surface water can be reduced to the 3- to 6-mg/L range, it is likely that surface water compliance with both acute and chronic aquatic criteria can be achieved.

6.7 Potentially Suitable Habitat

The Moab site is located within the 298-mile stretch of the Colorado River that has been designated as “critical habitat” for the Colorado pikeminnow. An additional 780 miles of the Green and the San Juan Rivers has also been designated critical pikeminnow habitat. In 1995, as part of a multi-agency effort to develop a plan for studying the effects of the Moab site on endangered fish habitat, the river was surveyed upstream and downstream of the site to identify suitable habitat areas for comparison. Six good pikeminnow habitats were identified: three upstream and three downstream of the site. An area adjacent to the site was also proposed for sampling even though the area was identified as “poor endangered species habitat (DOI 1995).” This corresponds to areas described in this SOWP as areas A, B, and C (Figure 4−23), which have been identified as being the most suitable pikeminnow habitat in the immediate vicinity of the site.

A comparison of river flow rates and periods of inundation of the west bank of the Colorado River near the Moab site indicates that, on average, water is contained in areas A, B, and C (Figure 4−23) for only a 9-day period each year. This conclusion is based on the current channel morphology, which is actually undergoing nearly continuous changes. However, the data strongly suggest that the potential habitat area on the west bank of the Colorado River is largely a floodplain feature and not the important in-channel backwater habitat that is critical to species survival. No habitat considered as being “desirable” or “preferred” has yet been identified in the Moab site vicinity.

Furthermore, it appears that the influence of site-related contamination on surface water quality has been limited. Most of the samples displaying highly elevated concentrations of ammonia do not represent river water samples in areas where aquatic organisms of concern would be expected. Moreover, many historical sample locations contained only inches of water or were cutoff pools. It is possible that aquatic receptors could be exposed to elevated levels of site constituents if they were close to the riverbank, but concentrations drop off rapidly with distance from shore due to mixing with river water. The extent to which the site poses an actual risk to endangered fish is unknown because sampling protocols to date have not focused on this evaluation. However, it does appear that only areas very near the shore are affected by site-related contamination, and the site has not had a negative effect on overall Colorado River water quality.

6.8 Tailings-Pile Management Alternatives

The range of reasonable surface remediation alternatives presented in the EIS (DOE 2003b) includes both on-site and off-site disposal of the tailings and contaminated soils. As a result, the analysis of ground water remediation alternatives includes site conditions under no-action, on-site, and off-site surface remediation alternatives. These three alternatives are described in the next sections.

6.8.1 No Action

Under the no-action alternative, there would be no remediation of the uranium mill tailings, surface soil contamination, vicinity properties, or the contaminated ground water. This alternative is included to provide a basis of comparison for the two action-based alternatives.

A simplified approximation of the liquid drainage rate and ammonia concentration that may result under the no-action alternative is presented Figure 6–2. The seepage rate of pore fluids from the tailings material is estimated to continue declining from the current rate of 20 gpm (18 gpm is apportioned for the sand tailings and 2 gpm is apportioned for the slimes tailings) until consolidation of the tailings is complete, and a steady-state condition of 8 gpm is reached after approximately 20 years. At that time, seepage from the sand tailings will be approximately 6 gpm, and seepage from the slimes tailings will be approximately 2 gpm. The seepage rate from the tailings as a function of time is summarized in Table 6–2.

Initially, seepage of pore fluids from the base of tailings pile is expected to contain a continuous source concentration of 1,100 mg/L ammonia. But the ammonia concentration of the leachate from the tailings pile is expected to increase as a precipitated salt layer starts to dissolve. The rate of dissolution and migration of the precipitated salt layer to the ground water is estimated by considering the thickness of the salt, its density, and solubility. From these parameters the mass of available salt per unit area is estimated to be approximately 30.5 grams per square centimeter (g/cm^2). Solubility of the salt is assumed to be approximately equal to the present “equilibrium” concentration (200 g/L) of the pore fluid in the upper 10 ft of the slimes facies. To dissolve the available salt at the equilibrium concentration (200 g/L) would require a specific volume of water of approximately 0.1524 L/cm^2. For the present infiltration rate of 1 H 10^{-7} cm/s, the salt layer would dissolve over a period of 48 years. Assuming a unit hydraulic gradient, moisture content of 0.25, and a travel distance of 70 ft from the salt bed to the water table, the salt would enter the ground water at 168 years from the present and be completely depleted at 217 years from the present. Seepage of pore fluids from the tailings pile is then assumed to continue indefinitely at 1,100 mg/L ammonia.

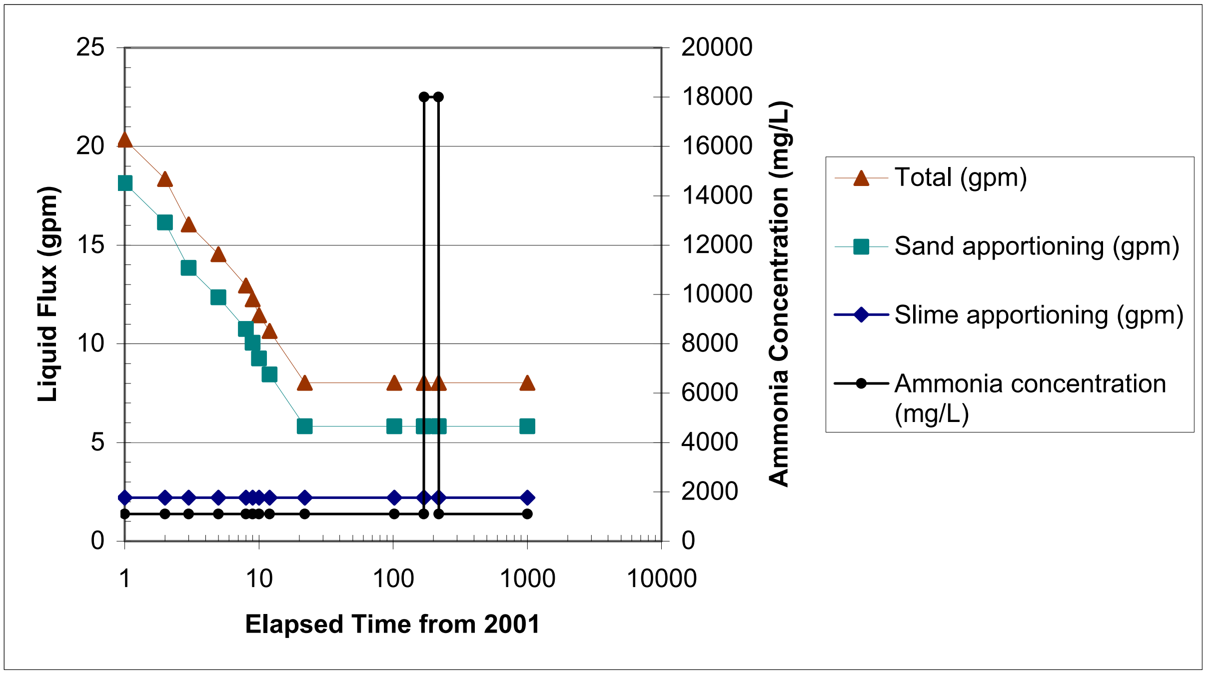


Figure 6–2. Summary of Liquid Drainage Rate and Ammonia Concentrations from the Tailings Pile for the No-Action Alternative

Table 6–2. Summary of Liquid Drainage Rate and Ammonia Concentrations From the Tailings Pile for the No-Action Site Conceptual Model

Parameter	Value
Infiltration Rate	1×10^{-7} cm/s
Gravity Drainage	Constant rate: 8 gpm
Transient Drainage	Rate decays from 12 gpm at present to 0 gpm at 20 years
Initial Ammonia Concentration Seepage from Base of Tailings Pile	1,100 mg/L
Breakthrough Ammonia Concentration from Upper Salt Layer	18,000 mg/L
Arrival time	168 years
Final Concentration	1,100 mg/L
Exit time	217 years

6.8.2 On-Site Disposal

The on-site disposal alternative would consolidate all contaminated soils and stabilize the 130-acre tailings pile in place in an above-grade disposal cell at its current location on the Moab site. A final cover would be designed to meet the requirements of EPA standards in 40 CFR 192, utilizing DOE's experience with other uranium mill tailings disposal cell covers. Flood protection would be constructed along the base of the pile and cover materials for radon attenuation and erosion protection would be imported to the site from suitable borrow areas. The final design would meet the requirements of disposal cells under EPA (40 CFR 192) and NRC (10 CFR 40, Appendix A) standards.

An approximation of the liquid drainage rate and ammonia concentration that may result under the on-site disposal alternative is presented in Figure 6−3. The seepage rate of pore fluids from the tailings material is estimated to decline from the current rate of 20 gpm to 0.8 gpm. During the first 20 years, the decline in the seepage rate follows the same curve as the forecasted decline for the no-action alternative. Drainage from the tailings pile continues to decline during the next 110 years, at which point a new steady state drainage rate is established. At that time, seepage from the sand tailings is assumed to be practically 0 gpm, and seepage from the slimes tailings is approximately 0.8 gpm. The seepage rate from the tailings as a function of time is summarized in Table 6−3.

Initially, seepage of pore fluids from the base of the tailings pile is expected to contribute a continuous source of 1,100 mg/L ammonia to the ground water system. But the ammonia concentration of the leachate from the tailings pile is expected to increase as the precipitated salt layer starts to dissolve. The rate of dissolution and migration of the precipitated salt layer to the ground water is estimated with the same procedures described in Section 6.8.1, except the engineered cover reduces the infiltration rate to 1×10^{-8} cm/s. Under this condition, the salt layer would dissolve over a period of 442 years. Assuming a unit hydraulic gradient, moisture content of 0.25, and a travel distance of 70 ft from the salt bed to the water table, the salt would enter the ground water at 1,094 years from the present and be completely depleted at 1,536 years from the present. Seepage of pore fluids from the tailings pile would then continue indefinitely with a concentration of 1,100 mg/L ammonia.

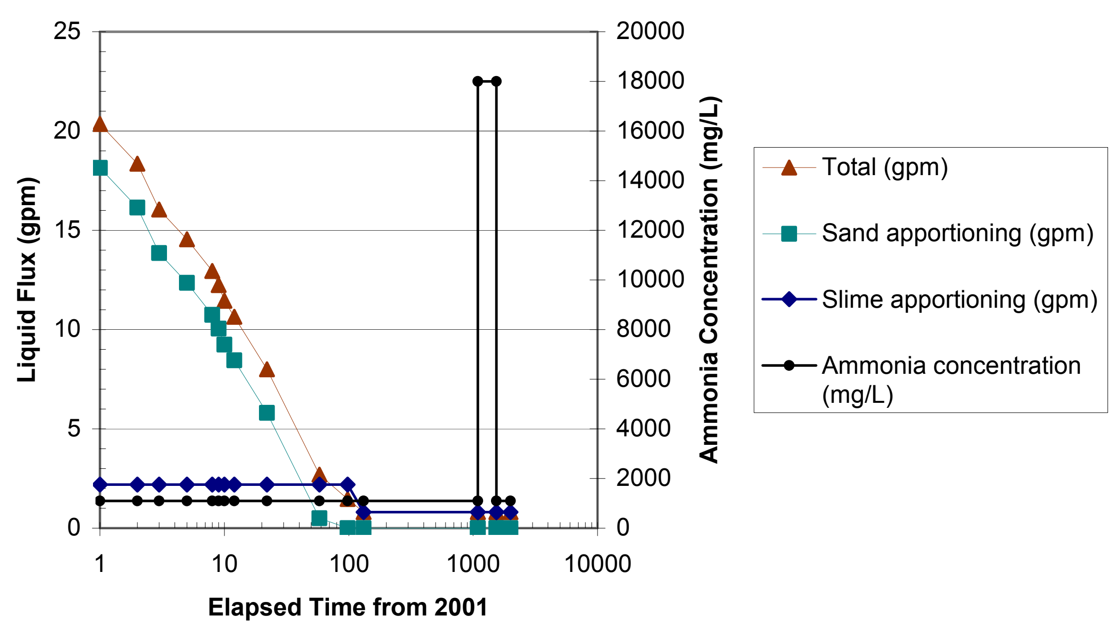


Figure 6–3. Summary of Liquid Drainage Rate and Ammonia Concentrations from the Tailings Pile for the On-Site Disposal Alternative

Table 6–3. Summary of Liquid Drainage Rate and Ammonia Concentrations From the Tailings Pile for the On-Site Disposal Alternative

Parameter	Value
Infiltration Rate	1×10^{-7} cm/s before construction and 1×10^{-8} cm/s after construction
Gravity Drainage	Rate decays from 8 gpm at present to 0.8 gpm at 132 years
Transient Drainage	Rate decays from 12 gpm at present to 0 gpm at 20 years
Initial Ammonia Concentration Seepage from Base of Tailings Pile	1,100 mg/L
Breakthrough Ammonia Concentration from Upper Salt Layer	18,000 mg/L
Arrival time	1,094 years
Final Concentration	1,100 mg/L
Exit time	1,536 years

6.8.3 Off-Site Disposal

For off-site disposal, the tailings pile and contaminated soils from the Moab site would be removed and transported to another location for disposal.

An approximation of the liquid drainage rate and ammonia concentration that may result under the off-site disposal alternative is presented in Figure 6–4. Seepage of pore fluids from the base of the tailings pile is expected to contribute a continuous source of 1,100-mg/L ammonia to the ground water system for approximately 9 years. At that time, it is assumed that the tailings pile is entirely transported to another location. The seepage rate of pore fluids from the tailings material is estimated to decline from the current rate of 20 gpm to 0 gpm by the year 2010. It is assumed that the source of ammonia contamination is entirely removed during that time. Dissolution and migration of the precipitated salt layer to the ground water would not occur under the off-site disposal alternative because the salt layer would be transported, together with the remaining tailings, to an off-site location for disposal. The seepage rate from the tailings as a function of time is summarized in Table 6–4.

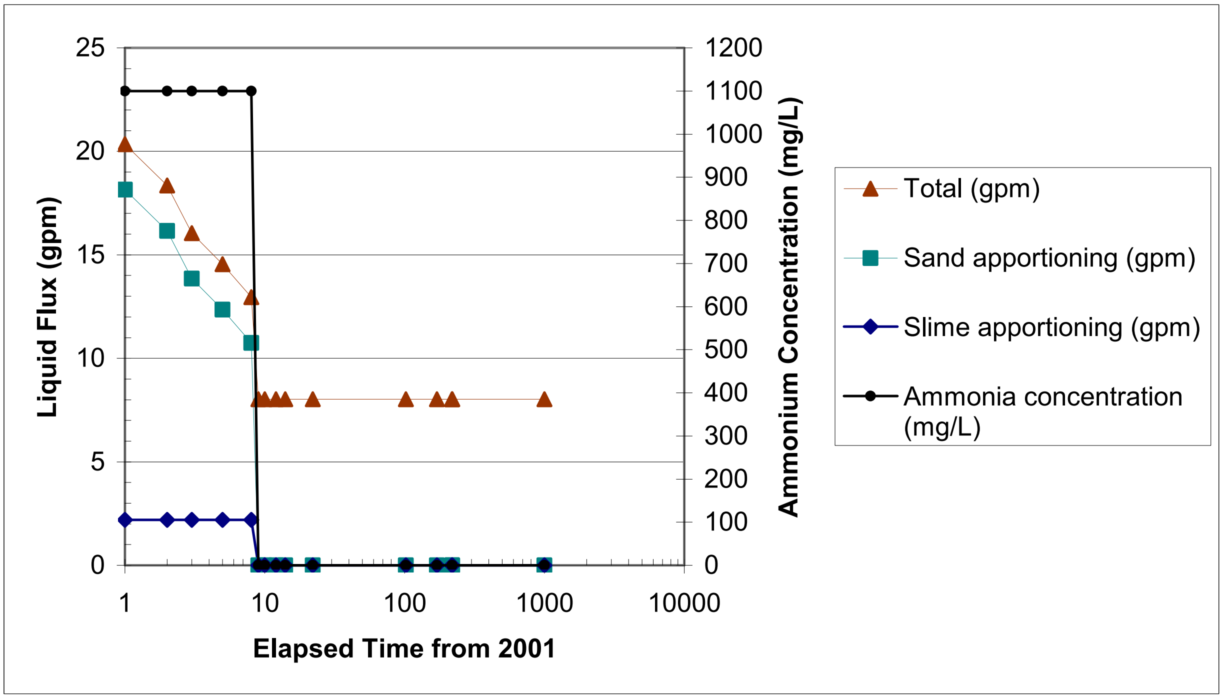


Figure 6–4. Summary of Liquid Drainage Rate and Ammonia Concentrations from the Tailings Pile for the Off-Site Disposal Alternative

Table 6–4. Summary of Liquid Drainage Rate and Ammonia Concentrations From the Tailings Pile for the Off-Site Disposal Model

Parameter	Value
Infiltration Rate	1×10^{-7} cm/s
Gravity Drainage	Constant Rate: 8 gpm
Transient Drainage	Rate decays from 12 gpm at present to 0 gpm at 9 years
Initial Ammonia Concentration Seepage from Base of Tailings Pile	1,100 mg/L
Breakthrough Ammonia Concentration from Upper Salt Layer	Not applicable
Arrival time	Not applicable
Final Concentration	0 mg/L
Exit time	Not applicable

7.0 Ground Water Flow and Transport Modeling

A flow and contaminant transport model was developed to evaluate the fate of ammonia contamination in ground water at the Moab site under three remedial alternatives for the tailings pile. The model was developed in accordance with the ASTM Standard Guide D5447-93 for *Application of a Ground-Water Flow Model to a Site-Specific Problem* (ASTM 1993). This section is structured following the ASTM Standard Guide D5718-*95* for *Documenting a Ground-Water Flow Model Application* (ASTM 1995).

7.1 Modeling Objectives and Scope

The function of the model was to predict the fate of ammonia in the contaminant plume as affected by several transport processes (advection, dilution, dispersion, sorption, etc.). Predicted concentrations of ammonia in the ground water adjacent to the Colorado River as a function of time were assessed to evaluate the potential impacts to aquatic species in backwater areas near the bank of the Colorado River.

The model accounted for contributions of ammonia from both the tailings pile and the brine zone beneath the saltwater interface. All available site data, such as groundwater levels in wells, aquifer tests, geologic logs, river levels, site chemical fluxes, and evapotranspiration study findings were used to construct and calibrate a 3-D, finite-element groundwater flow and contaminant transport model. The relative importance of individual flow and transport parameters affecting hydraulic head and contaminant distributions was evaluated through sensitivity analysis.

The modeling encompassed several tasks, including: (1) evaluating the hydrogeologic setting and developing a conceptual model, (2) selecting the code to be used in the analysis, (3) constructing the flow model, (4) calibrating the flow model, (5) constructing the transport model, and (6) performing predictive and sensitivity simulations. The hydrogeologic setting and site conceptual model are fully described in Sections 5 and 6, respectively, to which the reader is referred. The primary focus of this section is to document items 2 through 6 above and the archival of significant model logs and output files.

7.2 Code Selection

Major considerations in the selection of the computer simulator require that the model: (1) accurately represents the irregularly shaped geometry of the basin beneath the site, the slime tailings, the sand tailings, and the tamarisk area, and (2) be readily convertible to a 3-D, variable-density flow and transport simulator if salinity-affected flow and chemical transport near the saltwater interface were considered important. The following codes were examined for their abilities to meet these criteria:

MOCDENSE (Sanford and Konikow 1985)

MODFLOW/MT3D (McDonald and Harbaugh 1988)

MODFLOW SURFACT (HydroGeoLogic 1996)

SEAWAT (Guo and Langevin 2001)

SUTRA (Voss 1984)

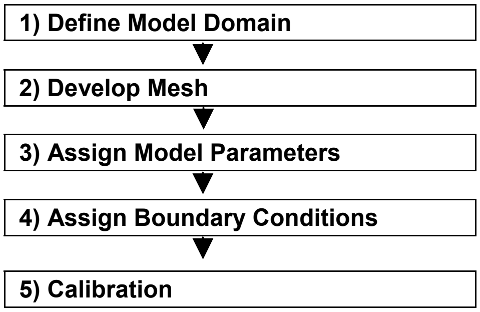
FEFLOW (Diersch 2002)

MOCDENSE, though capable of simulating variable density flow, was considered limiting because it could not represent 3-D flow characteristics of the site. The MODFLOW/MT3D combination was considered advantageous because of its universal application, but was rejected because of its inability to simulate variable density flow and transport. MODFLOW SURFACT has the capability to simulate variable density flow but, being a finite-difference code, is less suited to the irregular boundaries of the site. SEAWAT is a variable density code derived from MODFLOW/MT3DMS that can be applied using a popular graphical user interface. Though SEAWAT was found to be useful for two-dimensional modeling of cross-sectional density-affected flow and transport (Appendix D, Calculation X0062600), it has the same limitations that all finite-difference codes have in handling irregular boundaries. When the variable-density, finite-element simulator SUTRA was considered for this project, the 3-D upgrade was only available in a beta version. Ultimately FEFLOW was selected for the 3-D modeling of the site because it met all simulation criteria. It also has a "friendly" graphical user interface.

The finite elements available in FEFLOW are capable of accurately representing the irregular shape of the upper Moab valley, the source and sink areas in and near the site, and diverging flow observed in some portions of the valley covered by the site. Integral components of FEFLOW, such as interactive graphics, a GIS interface, data regionalization and visualization techniques allow for rapid input of data and evaluation of results. Robust solution techniques in the code allow the user to choose from various time-stepping techniques and matrix solvers. Transport processes simulated include advection, dispersion, and chemical reaction of contaminants in the ground water system. Boundary conditions in FEFLOW contain various options for evaluating the removal of chemical mass from the ground water system (Diersch 2002). These options were tested during the modeling process and some were utilized to achieve an acceptable mass balance. The FEFLOW code is fully described in the references cited and has been used as a comparison code in benchmarking studies described in the literature (Diersch 2002).

7.3 Ground Water Flow Model Development

This section presents the logic used to develop the 3-D finite-element groundwater flow model. The flow model was constructed in five sequential steps, as illustrated in the flow chart.



7.3.1 Define Model Domain

The model domain was selected on the basis of the local hydrogeology and the site conceptual model as previously presented in Sections 5.0 and 6.0, respectively. The model area in relation to the physical geography of the site is presented in Figure 5−17. The domain encircles the hydrogeological features that control flow and transport at the site. Included within the model domain are: (1) the Colorado River, Moab Wash, the Moab Fault zone, parts of the Glen Canyon Aquifer and Entrada Sandstone, and (2) the saturated zone extending from the potentiometric surface to either the bedrock or the saltwater interface. The locations of model boundaries were selected to prevent prescribed boundary conditions from inappropriately constraining the effects of internal stresses on the flow system.

7.3.1.1 Extent of Model Domain

The surface area of the model domain, as shown in Figure 5−17, is 4.8×10^6 square meters (m^2). The northeastern and southwestern boundaries of the model coincide with sharp bedrock slopes leading up to mesas in these areas, as defined by digital elevation data provided by the United States Geological Survey (USGS). The toes of the mesas are included in the model domain to account for potential local recharge from the bedrock. The southeastern model boundary was located approximately 1,000 ft east of the Colorado River to include a portion of Matheson Wetlands Preserve. The northwestern boundary was positioned in Moab Wash approximately 500 ft upstream of well RW-01, the most upgradient monitoring well.

The upper model surface was defined by the water table as interpreted from measured water levels at 38 monitor well locations in 2001 and 2002 (Section 5.2 and Figure 5−13). The bottom of the domain was defined either by the top of bedrock or the saltwater interface, both of which were assumed to comprise no-flow boundaries. Three-dimensional bedrock elevation and TDS data (Plates 2 through 10) were used to define the base of the model. The maximum thickness of the model domain is 135.07 meters (m). The volume of the 3-D model is approximately 1.5×10^6 cubic meters.

7.3.1.2 Model Domain Assumptions

In developing the model, it was assumed that all ground water discharge from the model domain occurs either as outflow to the Colorado River or evapotranspiration by phreatophytes. This conceptualization was also adopted by Sumsion (1971; p. 24-25; Plate 2), Blanchard (1990; p.31), Steiger and Susong (1997), and Eisinger and Lowe (1999; p.18), each of whom recognized the significance of the Colorado River as a regional ground water sink and phreatophytes as important consumers of ground water. Shallow ground water on either side of the Colorado River flows toward it as shown in Figure 5−13. These observations suggest that the Colorado River acts as a site of regional discharge. On the basis of the work reviewed above and the observations made to date (Section 5.2), it is plausible to assume that very little ground water, if any, flows under the river from one side to the other.

It is assumed that the northeast and southwest model borders comprise no-flow boundaries. This assumption appears feasible given the very low permeabilities that tend to dominate the bedrock units found along these borders.

7.3.2 Mesh Construction

Five steps were required to construct the model mesh. In the first step, model nodes were assigned to monitor well locations. The SEEPro database initially identified 159 monitoring wells at which ground water levels had previously been measured. Twenty-nine of these locations were found to lie outside the model domain. Node locations were not assigned to an additional sixteen because they shared the same coordinates as other monitoring wells. Ultimately, nodes were assigned to 114 monitoring locations shown in Figure 7–1.

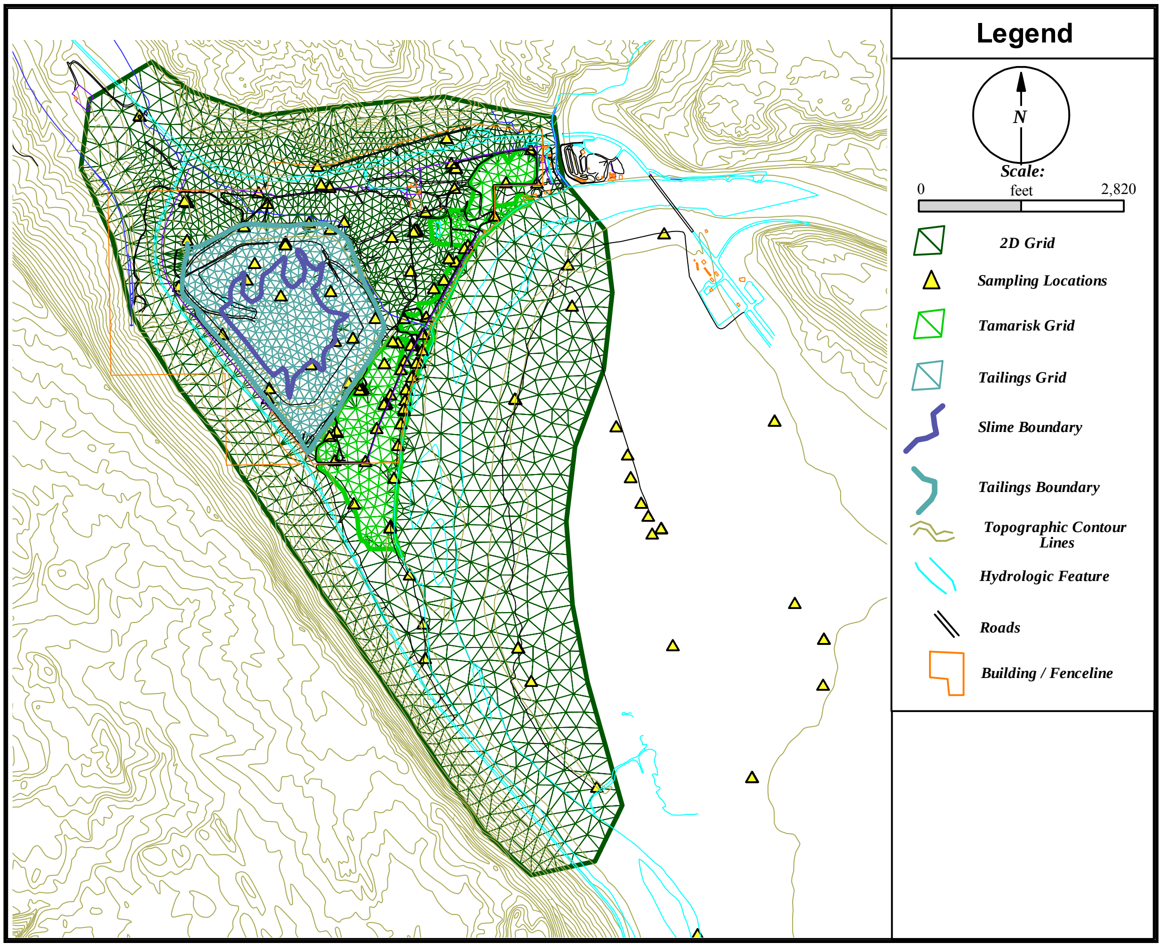


Figure 7–1. Finite Element Grid for the Moab Flow and Transport Model

In the second mesh development step, nodes were assigned to the boundaries of the tailings pile and along the borders of evapotranspiration zones (Section 5.2.3). Two separate footprints were identified within the tailings area on the basis of tailings material properties: a coarser-grained sand tailings area and a fine-grained slime tailings area (Section 5.4). The locations of the tailings footprints and the areas covered by the evapotranspiration zones are shown in Figure 7–2. Arc View shape files of these three zones were discretized and refined using the FEFLOW mesh generator. The resulting horizontal model mesh was composed of 5,717 finite element cells as shown in Figure 7–1.

The third step in construction of the model mesh involved the identification of three key horizons, including the uppermost and bottom model surfaces. The uppermost surface in the model represents the water table. The ground water contour map presented in Figure 5−13 was used to generate this surface of the model. The next surface in the model is the lithologic boundary that separates the upper fine grain alluvium, dominated by sand, from the lower coarser alluvium that is dominated by gravels. This lithologic contact is shown in cross-sectional views in Plates 2 through 10. The contour map presented in Appendix D, Calculation X0020900 showing the elevations along this surface was used to generate this model layer. The base of the model is either the top of bedrock or the salt-water interface, both of which are assumed to be no-flow boundaries. This surface was determined from top-of bedrock data presented in Plates 2 through 10 and reported TDS levels in ground water presented in Section 5.3.5.

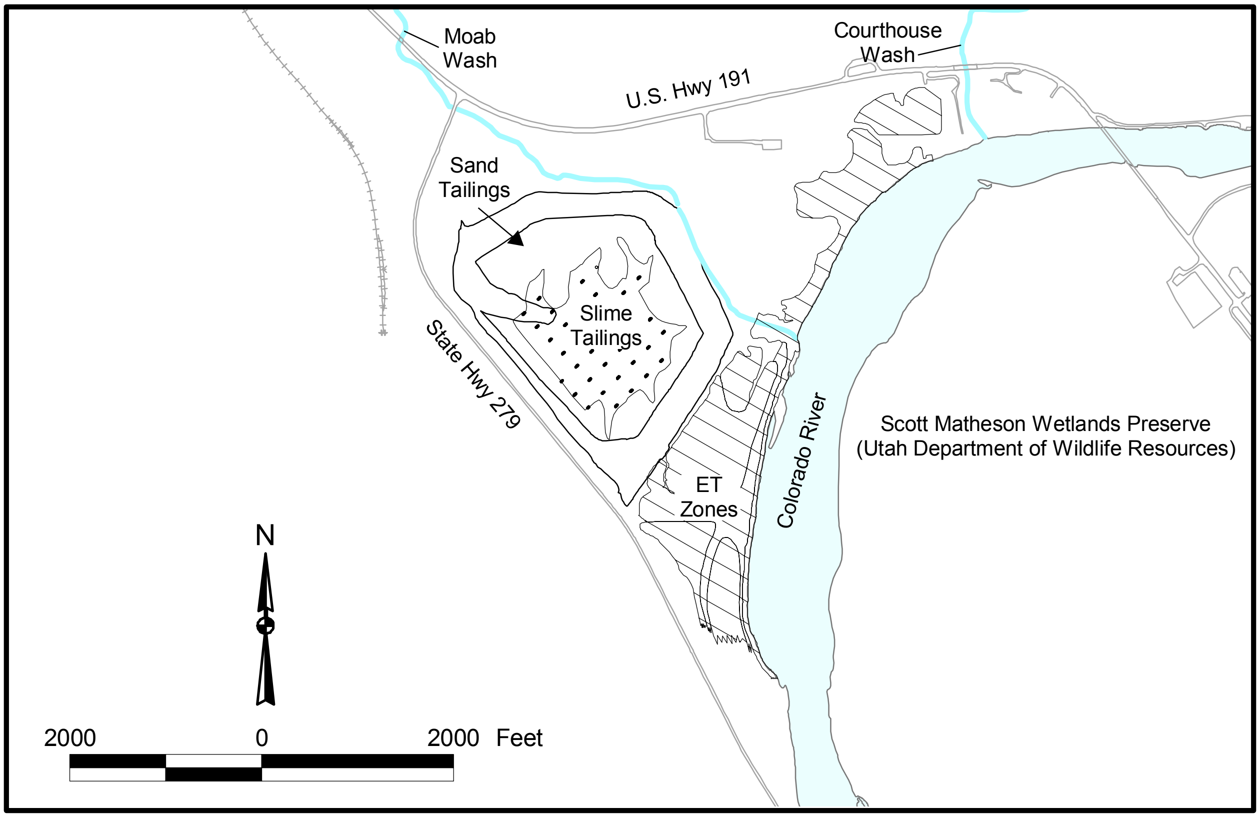


Figure 7–2. Distribution of Slime/Sand Tailings and Tamarisk Evapotranspiration Zones

The fourth construction step comprised vertical discretization. This was accomplished by dividing the full vertical extent of the model into 15 layers. (Layers in FEFLOW correspond to the vertical extent of the elements used to build the mesh; the top and bottom surfaces of the elements are referred to as slices). The thicknesses of the layers were determined as follows:

1) Six layers were generated to uniformly divide the vertical distance between the uppermost model surface (water table slice 1) and the top of the gravel surface (bottom of layer 6, slice 7).
2) Three thin layers were added above the base of the model: Layer 15 was 3.28 ft (1 meter) thick, and layers 13 and 14 were each 6.56 ft (2 meters) thick.
3) Six layers were added to uniformly divide the vertical distance between the gravel surface (bottom of layer 6, slice 7) and the top of layer 13 (slice 13).

The final step in constructing the model mesh involved local fining of the mesh both in the area of the tailings pile and the region underlain by the saltwater interface. The mesh was refined to improve both model stability and precision in areas where the total thickness of the model is decreasing.

7.3.3 Assigning Model Parameters

Initial values were developed for three types of porous medium properties: 1) hydraulic conductivity, 2) porosity, and 3) storativity. Hydraulic conductivities were later optimized during calibration.

7.3.3.1 Hydraulic Conductivity

Literature values of hydraulic conductivity (Freeze and Cherry 1979; p.29) were assigned on the basis of each soil classification, or soil type, logged within a borehole. The resulting hydraulic conductivity values used for the soil types encountered at the site are shown in Table 7–1. Of the 114 boreholes with available lithologic information, only 57 contained adequate information to assign hydraulic conductivity profiles. At these 57 boreholes, initial vertical and horizontal conductivity values were determined for 836 lithologic intervals.

Table 7–1. Initial Horizontal Hydraulic Conductivity Estimates Based on Different Soil Types Observed in Samples Collected at the Site

Soil Type	Hydraulic Conductivity (ft/d)	Hydraulic Conductivity (meters/second)
Clay	0.28	1.0×10^{-6}
Silt	2.8	1.0×10^{-5}
Clayey Silt	2.8	1.0×10^{-5}
Sand	14	5.0×10^{-5}
Gravel	140	5.0×10^{-4}
Siltstone	0.28	1.0×10^{-6}
Sandstone	0.28	1.0×10^{-6}
Gypsum	0.28	1.0×10^{-6}
Fill	2.8	1.0×10^{-5}

The assignment of hydraulic conductivities by model layer at each of the 57 wells having adequate lithologic logs was accomplished by calculating equivalent hydraulic conductivities. As provided in Freeze and Cherry (1979, equation 2.32), the equivalent horizontal hydraulic conductivity was calculated as

$$K_{xy} = \sum_{i=1}^{n} (K_i d_i) / d$$

Where K_{xy} = equivalent horizontal hydraulic conductivity (L/T),

K_i = horizontal hydraulic conductivity of the *i*th material (L/T),

d_i = thickness of the *i*th material (L),

d = model layer thickness (L),

n = the number of materials found in the model layer.

The equivalent vertical hydraulic conductivity was calculated using equation 2.31 in Freeze and Cherry (1979)

$$K_z = (d) / \sum_{i=1}^{n} d_i / K_i$$

where K_z = equivalent vertical hydraulic conductivity (L/T)

K_i = vertical hydraulic conductivity of the *i*th material (L/T)

In using the above equations, vertical hydraulic conductivities were assumed to be a tenth of horizontal values for each soil type.

The resulting computed equivalent horizontal and vertical hydraulic conductivities were subsequently interpolated over the FEFLOW model domain using the geostatistical method of *kriging*. Several intermediate steps were necessary to complete this interpolation. First, the equivalent horizontal and vertical hydraulic conductivities were converted to logarithmic space. The log-hydraulic conductivity data were then imported into Surfer (Golden Software), where they were kriged onto a finite-difference grid covering the same space as the finite-element model domain. The kriged log-conductivity data were then converted back to actual hydraulic conductivity values using the Fortran program *Invlog.for* (Attachment 1). This program rounded the Surfer-generated hydraulic conductivity values to the nearest half-order-of magnitude to account for the uncertainty associated with the hydraulic conductivity estimates. The Invlog.for program also assigned default values of hydraulic conductivity to any elements in the model domain that were located within bedrock. The hydraulic conductivity data were then imported layer-by-layer into FEFLOW, and transposed onto the finite-element mesh using inverse-distance interpolation techniques.

7.3.3.2 Anisotropy

As a result of applying the above-mentioned techniques for generating layer-by-layer hydraulic conductivities, the horizontal-to-vertical anisotropy of materials comprising the 3-D model ranged from 1:1 to 500:1. Bedrock materials were treated as isotropic (i.e., anisotropy ratio = 1). The largest anisotropy ratios were assigned to elements that were dominated by gravels but were also observed to contain numerous clay layers.

7.3.3.3 Porosity

Bedrock materials were assigned a porosity of twenty percent (20 percent) and alluvial materials assigned a porosity of thirty percent (30 percent) (Morris and Johnson 1967).

7.3.3.4 Storativity

As discussed in Section 4.9, estimates of aquifer storativity were derived from aquifer tests performed at ground water wells MOA-460, -461, -462, and –449. Application of the Hantush-Jacob Method (1955) to drawdown data at these wells produced alluvial materials storativities that ranged from 0.01 to 0.031 (Table 4−16). On the basis of these results, a uniform storativity of 0.02 was adopted in the FEFLOW model.

7.3.4 Assigning Boundary Conditions

Prescribed boundary conditions control the addition and removal of water in the model domain. Because the inflows and outflows along the outer portions of the model domain are not known with a high degree of confidence, the boundary conditions used to simulate them were treated as calibration parameters. Two general types of external boundaries are applied with the ground water flow model: (1) prescribed hydraulic heads and (2) prescribed flows, including no flow. Prescribed flows were used to represent recharge from precipitation, evapotranspiration, and inflows from tailings seepage.

7.3.4.1 Prescribed Heads

Hydraulic heads were prescribed along the northwest corner of the model to simulate inflow from Moab Wash close to where it exits Moab Canyon. Nodes associated with the top 6 model layers in this area were initially assigned a head of 3,963.25 ft above mean sea level as an approximation of locally observed ground water levels. Prescribed head conditions were also invoked to represent the Colorado River. The hydraulic heads used varied with distance along the river's course. Values for these spatially varied heads were calculated using inverse-distance interpolation of 9 river-elevations measured on May 20, 2002 (Section 4.4 and Appendix B). The locations and values of the May 2002 Colorado River elevations are shown in Figure 7–3.

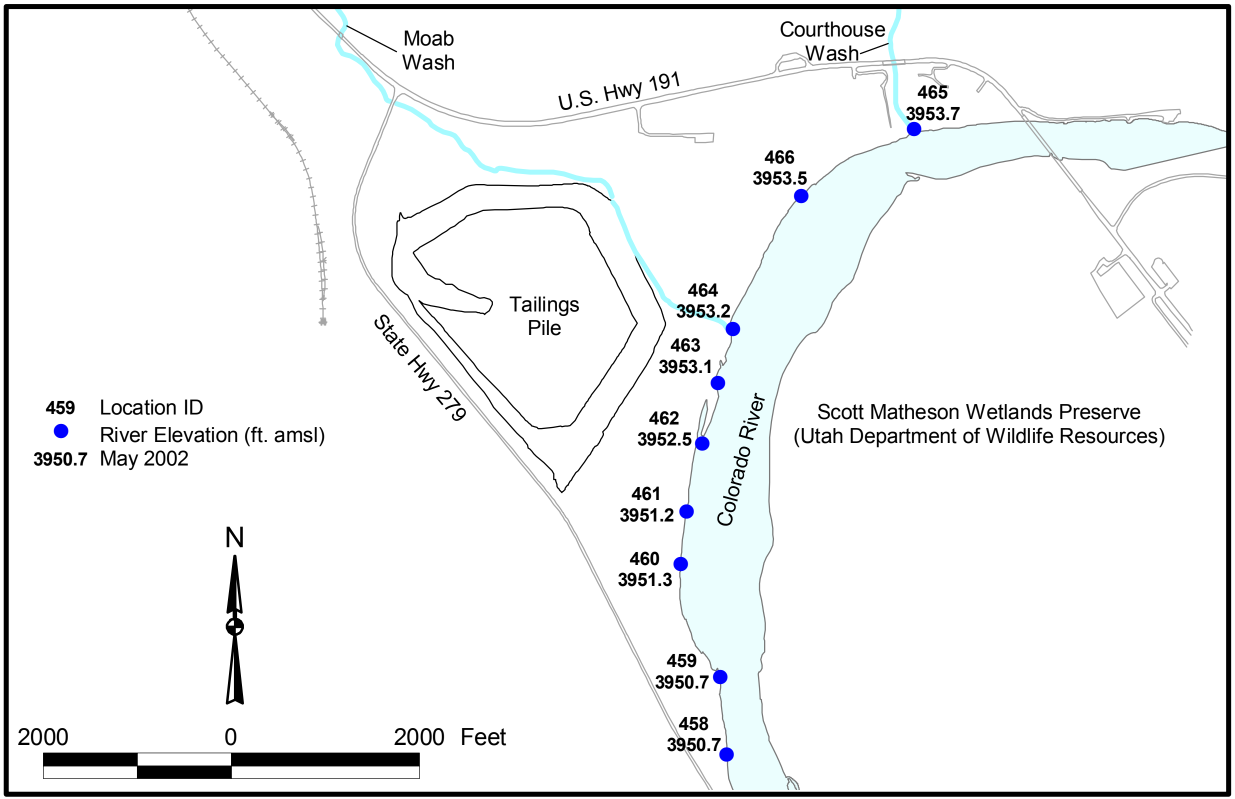


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Figure 7–3. Colorado River Surface Elevations Measured in May 2002

7.3.4.2 Prescribed Flows

Prescribed flow boundary conditions were used to represent four processes: (1) inflow from the Glen Canyon Group and Entrada Sandstone aquifers, (2) inflows from bedrock along the Moab Fault zone, (3) evapotranspiration from the tamarisk zones, and (4) inflows due to seepage from the tailings pile. Initial values for the prescribed flows were derived using a combination of parameters published in scientific literature, data from Moab site reports, and site-specific calculations.

Inflow from the Glen Canyon Group and Entrada Sandstone

Ground water discharge to the alluvial aquifer from Glen Canyon Group sediments, and to a lesser extent the Entrada Sandstone, was assumed to occur along the contact between alluvium and these bedrock units; the surface expression of this contact is shown in the geologic subcrop map presented in Plate 11. Head-calibration targets in the region of this boundary condition are sensitive to the magnitude of the prescribed flux. As described above, the prescribed values assigned to both of these boundaries were determined through calibration. After calibration was complete, a specific discharge of 0.012 meters per day (m/d) had been assigned to each node along the northern bedrock outcrop to layers 2 through 15 of the model.

Inflow from Moab Fault

Inflow from bedrock in the vicinity of the Moab Faults was simulated by assigning a prescribed specific discharge of 0.07 m/d to nodes along the base of the model domain in areas where the Moab Fault is in contact with either the Glen Canyon Group or the Entrada Sandstone. The location of the Moab Fault zone was obtained from the geologic map presented in Plate 2. The magnitude of the influx along the fault zone was arrived at through the model calibration process. This boundary condition was adopted on the premise that the majority of the water moving through the block of highly fractured bedrock aquifers would discharge adjacent to the fault where the effective vertical hydraulic conductivity of the bedrock is potentially at its largest value.

Outflow to Tamarisk

Tamarisk, or salt-cedar, is a small deciduous exotic tree introduced to the American Southwest from southern Eurasia (Horton 1977). Using infrared sensors, DOE identified three zones of tamarisk vegetation (Section 4.10.3) that were distinguished from each other by the degree of stress the plants are experiencing. The locations of the three tamarisk zones identified in the study are shown in Figure 4−27; the total area covered by tamarisk plants is shown in Figure 7–2. Initial estimates of the range of per-area ET rates (ET fluxes) for the three zones were developed using the results of tamarisk ET studies in the scientific literature. A summary of these estimated rates and the volumetric water losses resulting from applying them to the ET areas is presented in Table 7–2. The total water loss from ET was initially estimated to range from 208 to 504 gallons per minute (gpm).

Table 7–2. Evapotranspiration Rates Estimated for Tamarisk at the Moab Site

Tamarisk Zone	Flux (m/d)		Area		ET Rate (m^3/d)		ET Rate (gpm)	
	minimum	maximum	ha	m^2	minimum	maximum	minimum	maximum
1	0.007	0.014	13.2	132,000	904	1,808	166	332
2	0.003	0.007	8.3	83,000	227	568	42	104
3	0	0.003	13.5	135,000	0	370	0	68
Total	0.010	0.023	35	350,000	1,132	2,747	208	504

During early stages of model calibration, ET rates were assigned values that fell in the middle of the ranges of estimated rates listed in Table 7–2. However, calibration was best when tamarisk uptake rates near the low end of the listed ranges were applied. The ET rates used in the final calibrated model produced a total ET loss of about 150 gpm. The discrepancy between ET rates estimated on the basis of past work and the rates derived through modeling was considered a reflection of the uncertainties associated with this important component of the local hydrologic budget.

Since model calibration, information gathered by Waugh and van Reyper (2003) suggests that actual ET rates in the three tamarisk zones are about one-half of those listed in Table 7–2. Thus the total tamarisk uptake of 150 gpm used in the 3-D model is in good agreement with the more recent information.

Inflow from Tailings Pile and Slime Area

A prescribed inflow boundary condition was assigned to the area of the model underlying the tailings pile because drainage calculations (Appendix D, Calculation X0025700) indicated that saturated portions of the tailings pile continue to drain into the aquifer. Two separate model influxes were used to represent the current drainage rates from tailings; the area overlain by sandier tailings in the outer portions of the tailings was assigned a water flux of 2.5×10^{-4} meters/day, and the area overlain by slime tailings was assigned a flux of 6.9×10^{-5} meters/day. These rates represented volumetric inflows to the aquifer of 18 and 2 gpm, respectively.

The flux of pore fluids from the base of the tailings is expected to decline with time. A scenario simulated in the 3-D model has the total drainage rate gradually decreasing over the next 130 years from an initial 20 gpm to a steady-state flow of 0.8 gpm. A decay function available in the FEFLOW code was used to simulate this transient behavior. Time-varying tailings drainage rates produced by the model decay algorithm, as compared to predicted rates, are shown in Figure 7–4. The model tends to slightly underestimate the drainage during the initial time period and slightly overestimate the drainage for the later time periods and at steady state. Overall, however, the model-produced rates are in good agreement with the fluid flux estimated in the tailings drainage calculation.

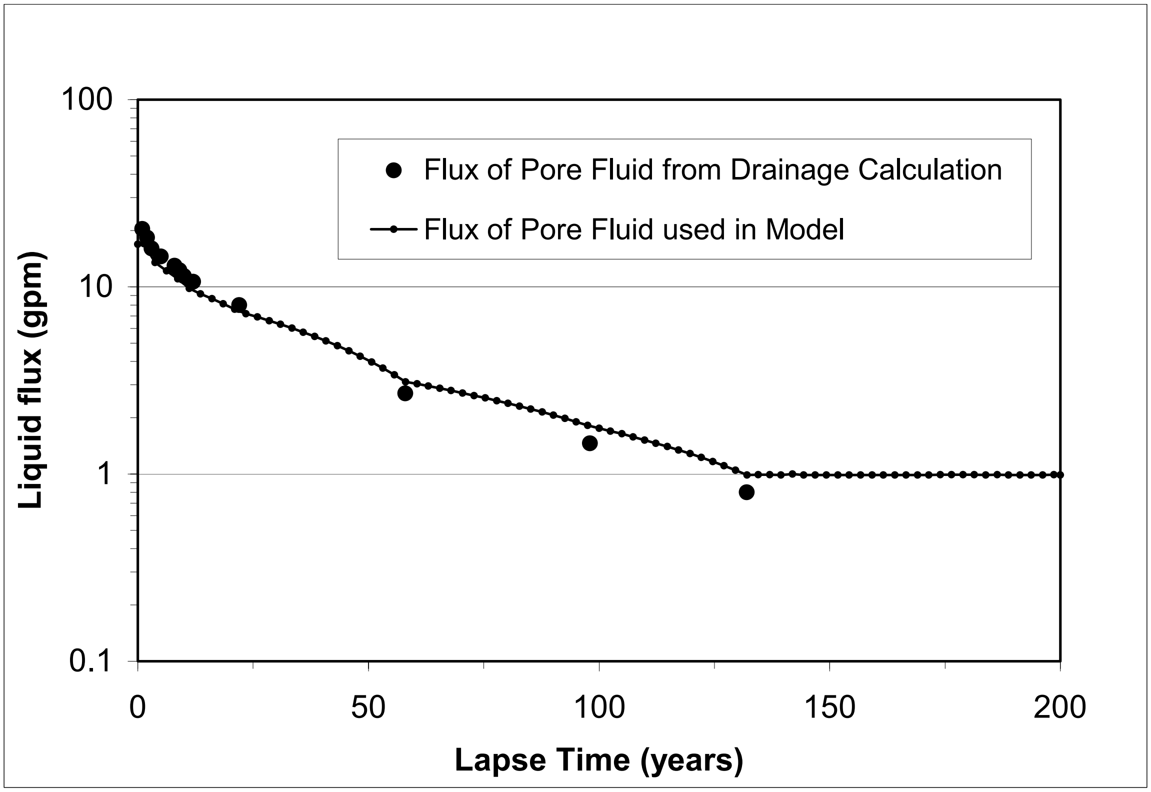

Figure 7–4. Comparison of the Flux of Pore Fluid from the Tailings Pile Estimated by the Drainage Calculation versus the Flux Simulated by FEFLOW

7.3.4.3 Recharge

Recharge from precipitation at the Moab site was also handled in the 3-D model with a prescribed flow boundary. The average annual precipitation was reported to be 9.09 inches per year (Section 5.2.3). Approximately 10 percent of this value is prescribed as inflow to the model as a constant flux that enters the model domain along its upper surface (model slice 1). The recharge is applied to all areas of alluvial outcrop except for the Colorado River and the tailings pile.

7.3.4.4 No-Flow Boundaries

The bottom of the model corresponds to either bedrock or the saltwater interface, the latter of which is defined by a TDS concentration of 35,000 mg/L. The location of the saltwater interface in the basin-fill aquifer is assigned as a no-flow boundary relative to the vertical movement of ground water. With the exception of the previously described areas of bedrock aquifer inflow, the contact between bedrock and alluvium is treated as a no-flow boundary. The saltwater interface is also assumed to comprise a no-flow boundary.

7.3.5 Flow Model Calibration

The 3-D finite-element ground water flow model was calibrated under the assumption that the flow system currently exists in a state of dynamic equilibrium. With this assumption, it could be further assumed that hydraulic heads measured in site wells during the past few years are representative of a steady-state system, and that these heads could be matched with steady-state versions of the model. Accordingly, a set of average hydraulic heads computed from water level measurements made between December 2001 and October 2002 was used as a model calibration target. Another target set consisted of measured heads in observation wells during aquifer tests conducted at the site. Most of the calibration was accomplished through adjustment of assigned hydraulic conductivities and changes in boundary condition parameters.

7.3.5.1 Hydraulic Conductivity Adjustment

The final hydraulic conductivity field in the model was mostly developed by matching model-predicted drawdowns to measured drawdowns in observation wells during ground water aquifer tests. Four aquifer tests, involving a total of nine observation wells, were all conducted in predominantly gravelly sediments comprising the alluvial aquifer. The model was used to simulate each test individually. Horizontal and vertical hydraulic conductivities were scaled up or down, depending on the fit between simulated and observed drawdowns. This process was repeated several times until the difference between predicted and observed drawdowns at all wells used in the analysis were evenly distributed about zero to the extent practicable. The adjustment procedure required kriging of the hydraulic conductivity field for each layer during each iteration. The wells used for this analysis and a listing of observed drawdowns along with comparable drawdowns computed by the final calibrated model are presented in Table 7–3.

Table 7–3. Hydraulic Conductivity Calibration Results

Pumping Well Name	Pump Rate (gpm)	Pump Interval (hours)	Model Slice (Number)	Observation Well	Observed Drawdown (ft)	Simulated Drawdown (ft)	Difference between Simulated and Observed Drawdown (ft)
ATP-2-S[a]	20	6	9	OW-3	0.54	0.56	0.02
			9	OW-4	0.36	0.51	0.15
PW01[b]	78/58	24.2	12	SMI-PZ1M	1.95	0.86	-1.09
			14	SMI-PZ1D2	1.11	0.66	-0.45
PW02[b]	61	22.7	13	SMI-PZ2M1	0.98	1.09	0.11
			13	SMI-PZ2M2	1.00	0.99	-0.01
			16	PZ2D	0.91	0.98	0.07
PW03[b]	71	21	9	SMI-PZ3M	2.26	1.07	-1.19
			12	SMI-PZ3D2	0.58	0.72	0.14

[a] ORNL 1998
[b] SMI Inc. 2001

Hydraulic Conductivity Distribution

The final hydraulic conductivity distribution resulting from attempts to match aquifer test results is consistent with the hydrostratigraphy of the model domain. A site-wide perspective of the final hydraulic conductivities is shown in the fence diagram in Figure 7–5. General trends observed in this distribution include relatively low hydraulic conductivities in the uppermost six model layers used to represent silty-sand materials and the uppermost part of the alluvial basin fill, and the model's highest hydraulic conductivities in the underlying gravelly unit of the basin fill aquifer. Moab Wash alluvium, which comprises fine-grained sand, gravelly sand, and detrital material mostly in the northwestern portion of the site, also tends to exhibit larger hydraulic conductivities.

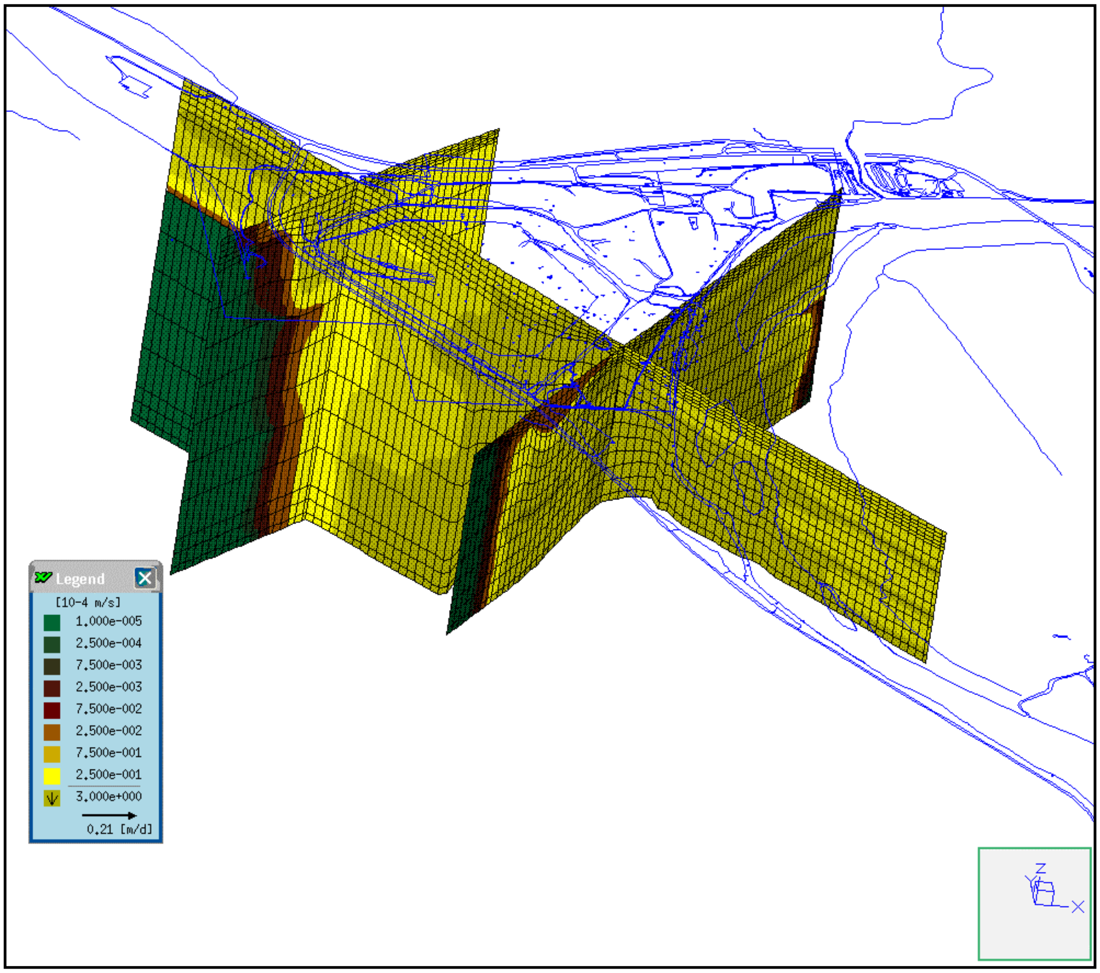

Figure 7–5. Fence Diagram Showing Vertical Distribution of Hydraulic Conductivity Throughout Model Domain

7.3.5.2 Boundary Condition Refinement

Final calibration of the model was achieved by adjusting fluxes along prescribed flow boundary surfaces (bedrock aquifer/alluvium contacts, including near the Moab Fault; recharge surfaces; ET zones). Trial-and-error techniques were employed during this phase of the calibration to match the observed heads to the model-generated heads.

7.3.5.3 Calibration Performance

Metrics used to evaluate the success of model calibration efforts were based on quantitative descriptors of the differences between hydraulic heads simulated by the ground water flow model (computed heads) and water levels measured in site wells (observed heads). These differences, which are referred to as residuals, were calculated at 38 different monitor well locations. Observed heads at these wells consisted of the average water levels measured at each location between December 2001 and October 2002.

Calibration Goal

A defensible goal for model calibration depends on the range of heads observed over the model domain. If the ratio of either the root mean squared error (RMSE) of model residuals or the standard deviation of residuals to the total range in head is small, the residuals comprise only a small part of the overall model response (Anderson and Woessner 1992; p.241). It was the goal of this modeling effort to achieve a ratio of standard deviation of residuals to the range in observed heads that had a value of 0.10 (i.e., 10 percent) or less. This ratio was calculated after each calibration run with the model using an Excel spreadsheet.

A map showing the residuals resulting from the final calibrated model is presented in Figure 7–6 A well-by-well listing of the residuals is also presented in Table 7–4. Analyses of these data indicate that the ratio of standard deviation of residuals to the observed head range is 0.055, or 5.5 percent, which meets the calibration goal.

A perfect model calibration produces a scatter plot of computed and observed heads wherein all data points fall on a diagonal line. A linear regression analysis of observed and computed heads in such a model would produce a coefficient of determination (R^2) of 1.0. Figure 7–7, which contains a scatter plot of computed and observed heads for the 3-D model, suggests that the calibrated model performs reasonably well in matching observed water levels in the 38 target wells. Linear regression analysis of the data used to construct this plot results in a R^2 value of 0.93, which signifies good correlation between computed and observed heads.

Ideally, the numerical model should not exhibit any inherent bias. Limited bias means that the arithmetic mean of the residuals should be as close to a value of zero as possible, and the residuals should be fairly evenly distributed above and below zero. A graph of model residuals versus observed heads, presented in Figure 7–8, indicates that the residuals have a slight negative bias, signifying that model-computed heads tend to be larger than the observed heads.

7.3.5.4 Summary of Calibrated Flow-Model

The model calibration process resulted in a total of six categories of horizontal hydraulic conductivity and eight categories of vertical hydraulic conductivity. As previously discussed, the largest inflows to the model were associated with prescribed flow boundaries used to represent bedrock aquifer discharge, both along the Moab Fault zone and in areas of contact between alluvium and the underlying Glen Canyon Group and Entrada Sandstone. The rate of inflow from areal recharge in the final model had a value of approximately 0.8 inches per year. Prescribed head boundaries were invoked along the Colorado River in model slices 1 and 2 to account for river/aquifer exchange, and in slices 1 through 6 near the Moab Canyon outlet to simulate inflows from Moab Wash alluvium.

A summary of the model-derived water budget for the calibrated flow model is presented in Table 7–5, both on a layer-by-layer basis and for the model as a whole. The summary shows that most inflow to the model occurs via subsurface sources, and most outflow occurs from the model’s shallowest slices.

Simulated hydraulic heads are contoured and presented in Figure 7−9. Observed hydraulic heads are contoured and presented in Figure 5−13 (Section 5.2) for comparison. Ground water contours predicted by the model closely resemble the observed.

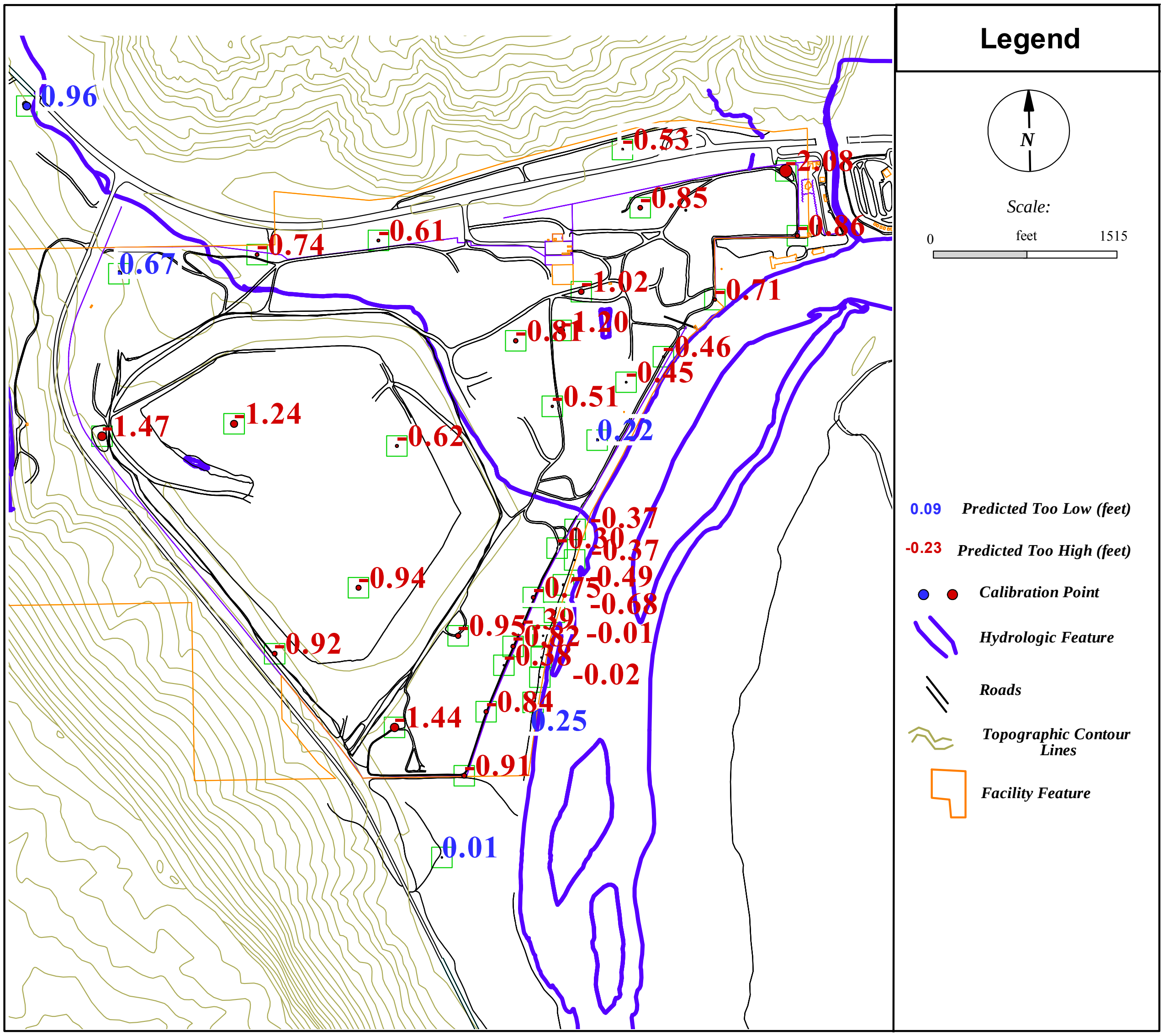

Figure 7–6. Posted Residuals at Target Locations for Calibrated Flow Model

Table 7–4. Calibration Target Residuals

Well Name	Observed Head (ft)	Predicted Head (ft)	Residual (ft)
401	3,953.57	3,953.18	-0.39
402	3,952.99	3,952.98	-0.01
403	3,952.69	3,952.67	-0.02
404	3,954.00	3,953.32	-0.68
405	3,953.92	3,953.55	-0.37
406	3,954.15	3,953.78	-0.37
407	3,952.23	3,952.48	0.25
409	3,953.96	3,953.47	-0.49
410	3,955.77	3,954.75	-1.02
411	3,956.05	3,955.24	-0.81
412	3,954.53	3,954.08	-0.45
413	3,955.17	3,954.66	-0.51
414	3,953.87	3,954.09	0.22
437	3,958.14	3,956.90	-1.24
438	3,956.45	3,955.83	-0.62
439	3,956.21	3,955.27	-0.94
440	3,958.82	3,957.35	-1.47
442	3,956.27	3,955.35	-0.92
443	3,959.76	3,960.43	0.67
455	3,957.84	3,957.23	-0.61
456	3,955.57	3,955.04	-0.53
457	3,955.33	3,954.48	-0.85
AMM-1	3,955.51	3,953.43	-2.08
AMM-2	3,954.38	3,953.63	-0.75
AMM-3	3,954.83	3,953.39	-1.44
ATP-3	3,959.95	3,959.21	-0.74
OW-1	3,954.87	3,953.92	-0.95
RW-01	3,962.04	3,963.00	0.96
SMI-PW02	3,953.59	3,953.21	-0.38
SMI-PZ1S	3,954.12	3,953.82	-0.30
SMI-PZ3S	3,956.04	3,954.84	-1.20
TP-01	3,954.44	3,953.73	-0.71
TP-02	3,954.32	3,953.86	-0.46
TP-07	3,953.24	3,952.33	-0.91
TP-08	3,953.65	3,952.81	-0.84
TP-09	3,954.15	3,953.33	-0.82
TP-11	3,954.37	3,953.51	-0.86
TP-20	3,951.78	3,951.79	0.01
Residual mean (ft)			-0.60
Absolute residual mean (ft)			0.71
Minimum residual (ft)			-2.08
Maximum residual (ft)			0.96
Range in observed head (ft)			10.26
Standard Deviation (ft)			0.57
Standard Dev/Range in Measured Heads (%)			5.53

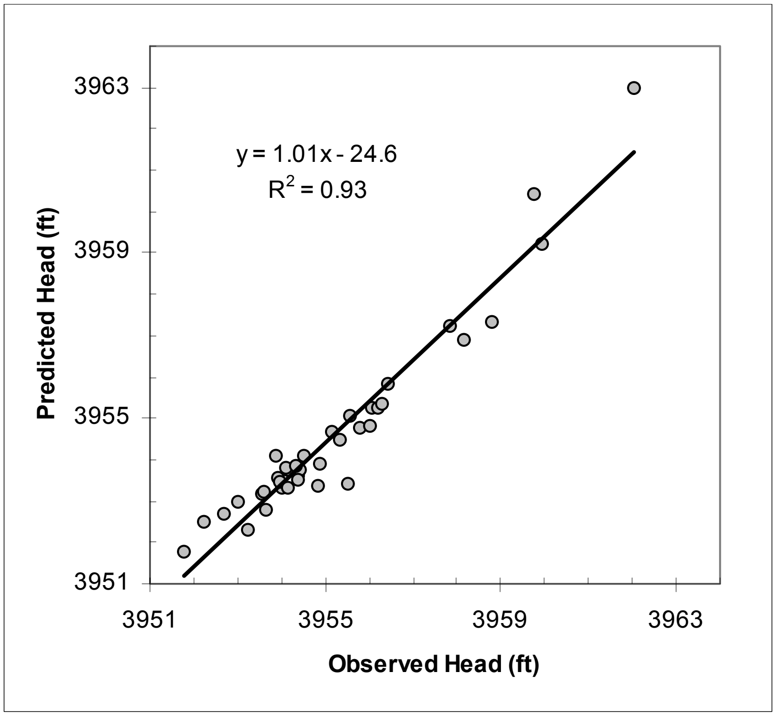

Figure 7–7. Comparison of Predicted vs. Observed Heads

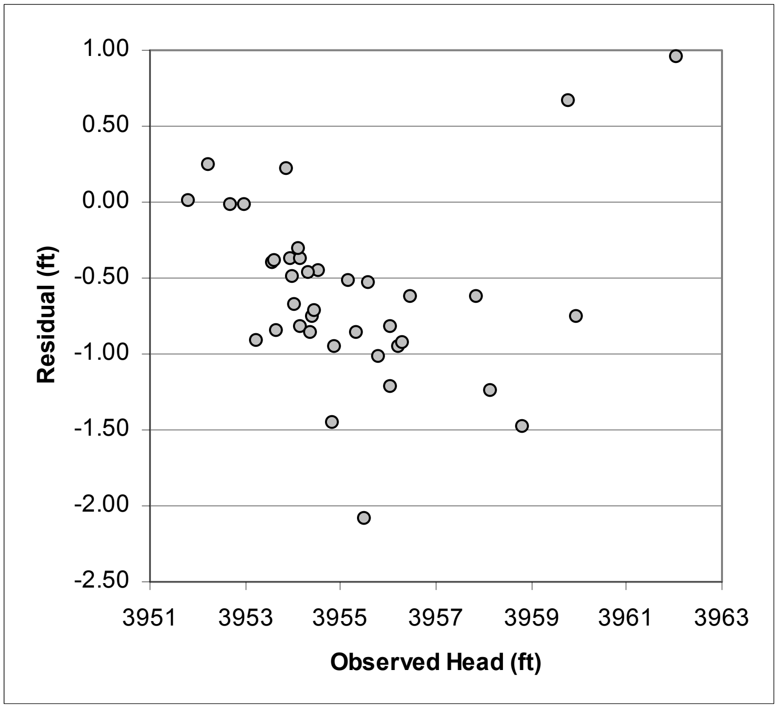

Figure 7–8. Comparison of Residual vs. Observed Head

Table 7–5. Fluid Balance for Calibrated Flow Model

Slice	Moab Wash		Tailings Pile	Colorado River		ET	Glen Canyon and Entrada	Moab Fault zone	Areal Recharge	Total		
	Inflow (gpm)	outflow (gpm)	inflow (gpm)	inflow (gpm)	outflow (gpm)	outflow (gpm)	inflow (gpm)	(inflow)	inflow (gpm)	inflow (gpm)	Outflow (gpm)	Imbalance (gpm)
1	0.0	0.0	0.0	0.0	1.4	18.6	0.0	0.0	26.3	26.3	20.0	6.4
2	0.1	0.0	17.0	4.6	273.1	46.2	0.6	0.0	0.0	22.3	319.3	-297.0
3	0.1	0.0	0.0	0.0	0.0	42.8	1.2	0.0	0.0	1.3	42.8	-41.5
4	0.1	0.0	0.0	0.0	0.0	40.9	1.2	0.0	0.0	1.3	40.9	-39.6
5	0.1	0.0	0.0	0.0	0.0	0.0	1.2	0.0	0.0	1.3	0.0	1.3
6	0.5	0.2	0.0	0.0	0.0	0.0	7.9	0.0	0.0	8.3	0.0	8.3
7	0.0	0.0	0.0	0.0	0.0	0.0	14.5	0.0	0.0	14.5	0.0	14.5
8	0.0	0.0	0.0	0.0	0.0	0.0	14.5	0.0	0.0	14.5	0.0	14.5
9	0.0	0.0	0.0	0.0	0.0	0.0	14.5	0.0	0.0	14.5	0.0	14.5
10	0.0	0.0	0.0	0.0	0.0	0.0	14.5	0.0	0.0	14.5	0.0	14.5
11	0.0	0.0	0.0	0.0	0.0	0.0	14.5	0.0	0.0	14.5	0.0	14.5
12	0.0	0.0	0.0	0.0	0.0	0.0	14.5	0.0	0.0	14.5	0.0	14.5
13	0.0	0.0	0.0	0.0	0.0	0.0	9.9	0.0	0.0	9.9	0.0	9.9
14	0.0	0.0	0.0	0.0	0.0	0.0	5.3	0.0	0.0	5.3	0.0	5.3
15	0.0	0.0	0.0	0.0	0.0	0.0	3.9	0.0	0.0	3.9	0.0	3.9
16	0.0	0.0	0.0	0.0	0.0	0.0	1.3	254.3	0.0	255.6	0.0	255.6
Total	0.9	0.3	17.0	4.6	274.5	148.5	119.7	254.3	26.3	422.7	423.0	0.3

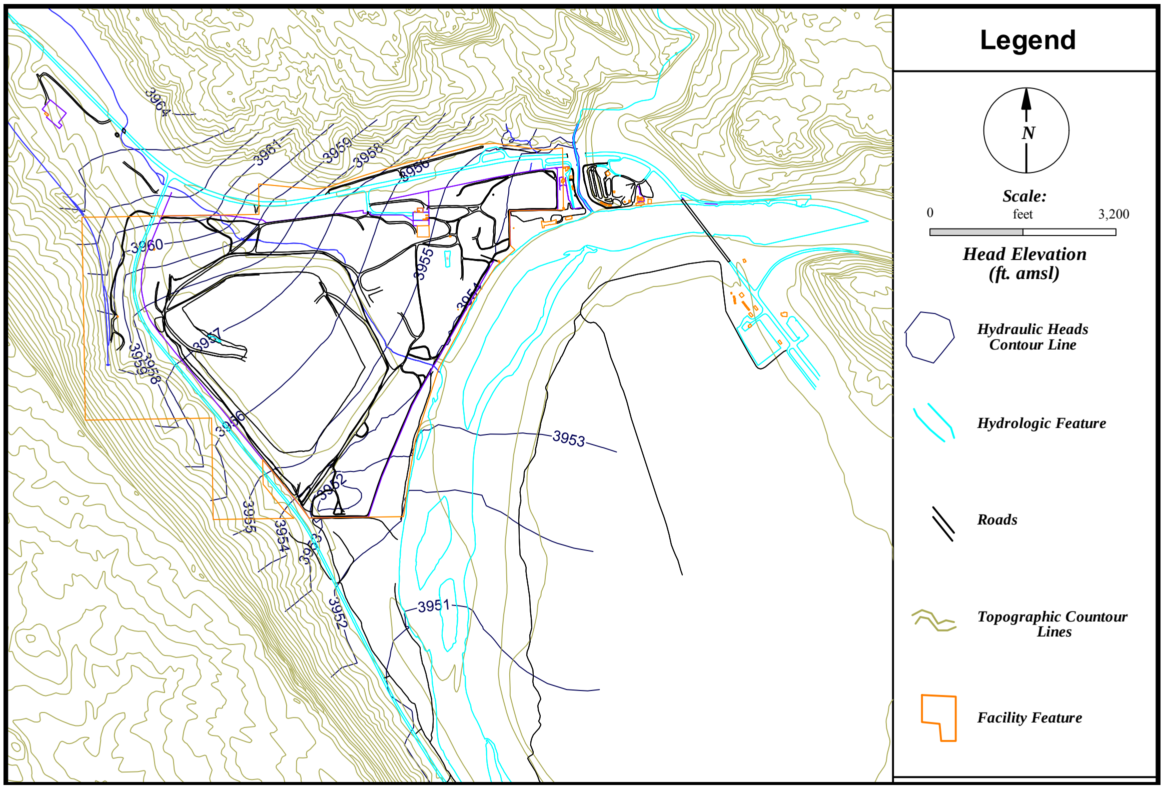

Figure 7–9. Contour Map of the Simulated Hydraulic Heads in the Calibrated Flow Model

7.3.5.5 Sensitivity Analysis

A sensitivity analysis was performed with the calibrated model to assess the relative influence of flow parameters and boundary conditions on computed hydraulic heads. Parameters varied during the sensitivity analysis included gravel hydraulic conductivity, prescribed inflows from the Glen Canyon and Entrada Sandstone aquifers, ET outflux, and Moab Fault influx. Impacts of varying these model parameters and boundary conditions individually are shown in Figure 7–10. The results indicate that the ground water flow model is most sensitive to gravel hydraulic conductivity and boundary flux from bedrock aquifers in the vicinity of the Moab Fault. Gravel hydraulic conductivities on the low end of the range assigned to this parameter produce poor calibration metrics, i.e., the ratio of residual standard deviation to the range of observed heads exceeds 10 percent. The 10 percent criterion is also exceeded at both the high and low ends of the sensitivity range for Moab fault prescribed influx. Regardless of the values assigned to influx from the Glen Canyon Group and Entrada Sandstone aquifers in the sensitivity analysis, the 10 percent criterion is always met. Figure 7–10 shows that, with the exception of tamarisk ET rates, the parameter values used in the final calibrated model consistently yield minimum model error.

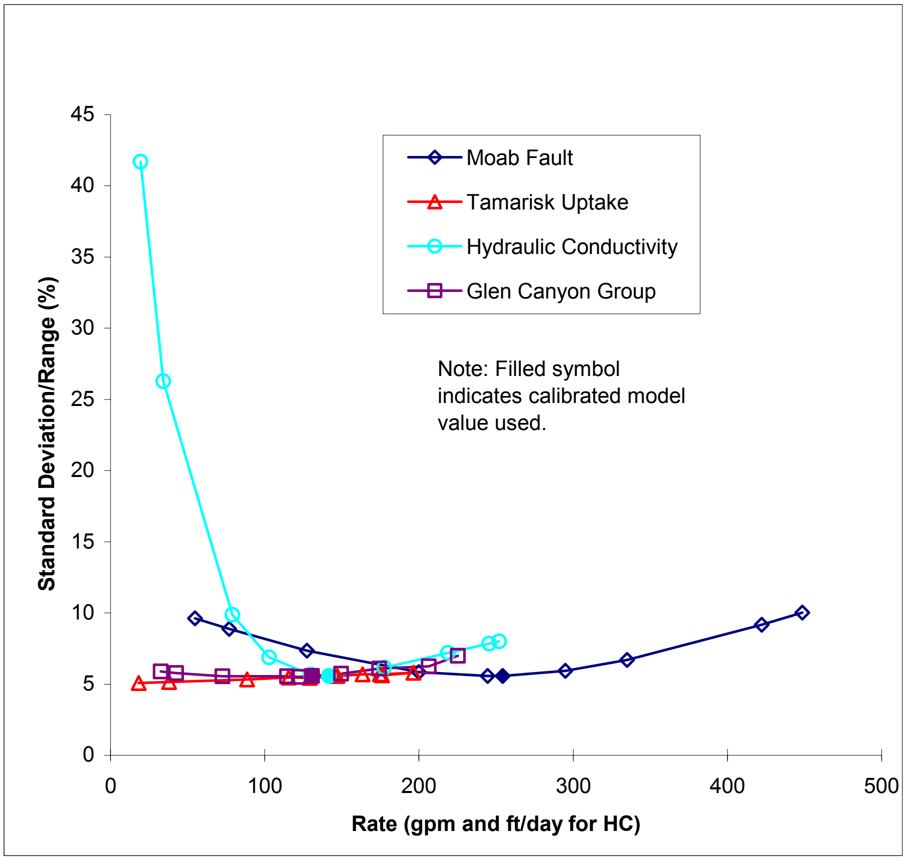

Figure 7–10. Single Parameter Sensitivity Analysis for the Moab Flow Model

Monte Carlo simulations were also performed with the 3-D model to identify parameter combinations that yield better calibration metrics. Three model inputs were treated as uncertain in the Monte Carlo simulations: Moab Fault influx, Glen Canyon and Entrada Sandstone aquifer influx, and gravel hydraulic conductivity. Results of the analysis, as presented in Figure 7–11, suggest that combinations of boundary influxes other than those adopted in the final calibrated model can produce better model calibration. The Monte Carlo results also indicate that the best calibration results are achieved using relatively large gravel hydraulic conductivities, i.e. conductivities that range from 5 x10^{-4} to 10 x10^{-4} meters per second (m/s) (140 to 280 ft per day [ft/d]). As with the single-parameter sensitivity analysis (Figure 7–10), gravel hydraulic conductivities less than the low end of this range tend to yield less favorable calibration metrics. It is noteworthy to point out that the calibrated model makes use of a gravel hydraulic conductivity of 5 x10^{-4} m/s (140 ft/d), which falls within the range exhibiting the best calibration results.

7.4 Transport Model Construction

The transport model was developed to evaluate the fate of ammonia contamination in the alluvial aquifer in response to three remedial alternatives for the tailings pile. Sources of ammonia contamination in the freshwater portion of the aquifer were accounted for in the transport model by simulating chemical mass influxes from the tailings pile and legacy ammonia contamination in the brine zone.

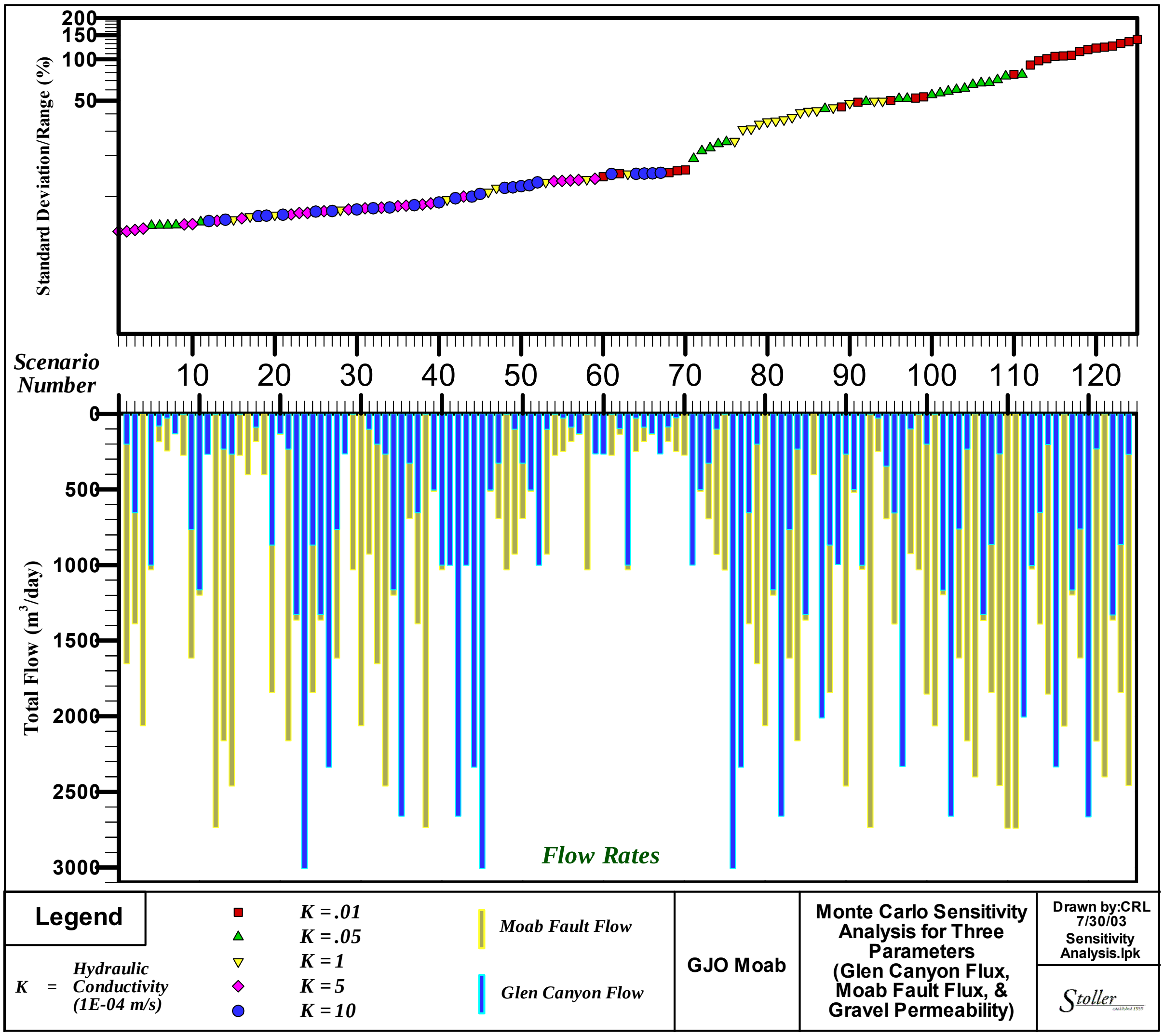

Figure 7–11. Multiple Parameter Sensitivity Analysis for the Moab Flow Model

Concentrations of several other site contaminants in ground water at the Moab site exceed their respective EPA ground water standards (Section 5.5). Although no EPA drinking water or ground water standard exists for ammonia, it was selected for evaluation because it is present in the tailings seepage and ground water at concentrations significantly greater than natural background and it is the key constituent driving the proposed ground water remedial action presented in the EIS due to its high concentrations discharging to the Colorado River and its associated toxicity to aquatic organisms. It is assumed that if ammonia target goals could be achieved that are acceptable for protection of aquatic life, concentrations of the other contaminants would also be protective. Even though the geochemical behavior of the other contaminants of potential concern differs from that of ammonia, it is anticipated that these constituents would be protective in the same time frame that it would take for ammonia to reach protective levels because they are less elevated above applicable cleanup criteria (e.g., surface water standards), are less widespread, or occur at elevated concentrations less frequently. For this reason, ammonia is the focus of the transport model evaluation.

Advection and mechanical dispersion of ammonia are dependent on pore-water velocities of ground water as computed in the site flow model. Advection comprises the transport mechanism whereby ammonia migrates at the same average rate with which water moves through aquifer pores. Dispersion is caused by variations in the paths of individual water molecules; it was accounted for with mechanical dispersion coefficients, each of which comprise the product of a model-computed pore-water velocity and a representative aquifer dispersivity (e.g., Freeze and Cherry 1979). Mechanical dispersion coefficients were added to molecular diffusion coefficients to produce hydrodynamic dispersion coefficients.

Dispersivity is very difficult to measure at field sites containing large plumes. Consequently, it is usually estimated using data and findings from the scientific literature. Researchers in the past have found that values of longitudinal dispersivity (i.e, dispersivity in the direction of the ground water hydraulic gradient) typically range from 2 to 30 percent of the length of a plume (e.g., Gelhar et al. 1992). For the Moab transport model, a longitudinal dispersivity of about 5 percent of plume length was adopted, resulting in a longitudinal dispersivity of 25 m. Transverse dispersivity was assigned a value of 2.5 m, which was 10 percent of the longitudinal value. Vertical dispersivity was assigned a value of 0.25 m.

Degradation of ammonia by nitrification is a first-order decay process. Short of having site-specific data regarding the rate(s) at which ammonia degrades, previous studies were consulted to ascertain reasonable decay rates for this constituent. Ultimately, a first order decay constant of 2×10^{-9} sec^{-1} was used. This value was identical to the ammonia decay constant adopted by Kipp (1987) for another site that bore only minimal resemblance to the Moab area. Despite the dissimilarity between Kipp's study area and the Moab site, use of this decay constant was found to produce and maintain distributions of ammonia in the 3-D model that were similar to those observed under current conditions.

Sorption of ammonia to aquifer sediments was assumed in the model to be a linear equilibrium process (Freeze and Cherry 1979) This means that the retarded migration of ammonia due to sorption could be simulated using a bulk transport parameter referred to as a soil/water distribution coefficient, or K_d. This parameter is typically used in conjunction with dry bulk density of the sediments comprising the porous medium and the medium's porosity to calculate a retardation factor.

A chemical's distribution coefficient can be approximated by a distribution ratio (R_d), a laboratory-derived parameter that is defined as the ratio of the concentration of a chemical in soil to its concentration in water under equilibrium conditions. Site-specific values of R_d for ammonia were determined using sediment samples from the alluvial aquifer (Section 4.7. Ultimately, an R_d value of 0.5 cubic centimeters per gram (cm^3/g) was adopted for all alluvial materials in the Moab model.

Contaminant retardation due to sorption is simulated in FEFLOW using a Henry isotherm (Diersch 2002). This method makes use of a chemical-specific Henry coefficient (k) and the density of solid material (sediment) comprising the porous medium (ρ_s). Using a ρ_s of 2.65 grams per cubic centimeter (g/cm^3) and the value of R_d mentioned above, the Henry coefficient employed in FEFLOW was calculated as follows:

$$R_d = (k)/(\rho_s)$$
$$k = (R_d)(\rho_s)$$

$$k = (0.5\ \text{cm}^3/\text{g})\ (2.65\ \text{g/cm}^3)$$
$$k = 1.325$$

7.4.1 Ammonia Concentrations

Ammonia contamination observed today in the ground water system at the Moab site is conceptualized as a legacy plume stemming from mill and tailings pile operation. From the perspective of the 3-D model, the dissolved ammonia remaining in ground water above the brine zone is now being fed by two sources: (1) downward seepage of contaminated fluids from the tailings pile, and (2) upward flux of legacy ammonia in the brine zone. A schematic of this conceptual model is shown in Figure 5−32.

The current distribution of the ammonia is described in Section 5 of the SOWP (Figures 5−28, 5−29, and 5−30). Initial concentrations were applied to the model and transport boundary conditions were used to account for the continuing impacts of these three sources.

7.4.1.1 Dissolved Ammonia Mass in the Alluvial Aquifer

The current distribution of ammonia in the alluvial aquifer is described in Section 5.5.1.1. Using this information, initial dissolved concentrations of ammonia were derived for each layer in the 3-D transport model. Under this process, which made use of kriging built into the EVS data visualization package, the total mass of dissolved ammonia assigned to the aquifer in the model was 1.17×10^9 grams. This value compared favorably with a previous calculation that put the total mass of dissolved ammonia in the freshwater portion of the alluvial aquifer at 1.06×10^9 grams.

7.4.1.2 Mass Flux from Tailings Pile

Tailings pore fluids were introduced into the model using a prescribed mass flux boundary condition. The *divergence* form of the transport equation (Diersch 2002) was solved within FEFLOW to assure that the assigned mass loading rates were maintained accurately.

Because three different remedial options regarding the tailings pile (no-action, on-site disposal, an off-site disposal) were examined with the model, it was necessary to develop three different sets of estimates for the ammonia mass loading from tailings fluid seepage. Section 6.9 describes how these chemical loading rates were estimated. Under all three remedial options, it was assumed that the average concentration of ammonia (1,100 mg/L) in the tailings seepage would remain constant, but that mass loading to the aquifer would decrease with decreases in the tailings seepage rate. The decline in seepage rates was taken from the results of a tailings-seepage model (Appendix D, Calculation X0025700). Separate seepage rates were estimated for the sand tailings that comprise the outer portion of the pile and the slime tailings comprising the pile's inner portion. Unique mass-transport functions were adopted in FEFLOW for these two general areas of tailings seepage.

7.4.1.3 Mass Flux from Brine Zone

A discussion of contaminant nature and extent in Section 5.5 describes how downward migration of tailings pore fluids during and subsequent to the time of milling is believed to have driven ammonia below the salt-water interface that currently defines the base of the 3-D model. The depth to which ammonia migrated downward and is observed in the brine zone today likely depended on the density of the tailings fluids during mill operation years. Under present conditions, the ammonia plume beneath the salt-water interface represents a long-term source of ammonia to the freshwater system. The locations where ammonia is believed to be entering the freshwater portion aquifer at the saltwater interface are depicted in Figure 7–12.

The rate at which ammonia in the brine zone will migrate into overlying brackish and freshwater was estimated using findings from 2-D density-dependent modeling performed for the site (Appendix D, Calculation X0062600). This modeling, which was conducted with the U.S. Geological Survey code SEAWAT (Guo and Langevin 2001) along a cross section originating in the northwest portion of the site in Moab Canyon and extending under the tailings pile to the middle of the Colorado River, indicated that the flux of ammonia across the saltwater interface would result from both advective and dispersive processes and would be transient in nature. The resulting influxes for a single location directly underlying the tailings were shown to decrease with time, as illustrated in Figure 7–13.

The 2-D density-dependent modeling (Appendix D, Calculation X0062600) indicated that the temporal decline in ammonia fluxes across the saltwater interface would vary depending on observed ammonia concentrations at the interface. To account for changes in ammonia influx due to concentration variations, the basic flux vs. time curve in Figure 7–13 (which was derived for one location) was adopted along other portions of the interface in a manner that accounted for local ammonia concentrations shown in Figure 7–12. Incorporation of these fluxes in the model was accomplished using a functional relationship that approximated the changes in flux with time (see Figure 7–13).

The density-dependent modeling also suggested that ammonia fluxes across the saltwater interface become more complex with proximity to the Colorado River, with many poorly quantified processes appearing to potentially affect these fluxes. Because of the considerable complexities associated with transport mechanisms occurring near the interface, no steps were taken in the model to explicitly account for them. Instead, attempts were made to account for possible variations in ammonia mass inflow caused by system complexities using robust uncertainty analysis techniques.

7.4.2 Summary of Transport Model Input Parameters

Transport model input parameters and boundary conditions were adjusted to yield a qualitative match between observed and simulated ammonia concentrations. Simulated ammonia concentrations for present conditions are shown in Figure 7–14. The qualitative match between simulated and observed ammonia concentrations at the water table was obtained by comparing Figure 7–14a to Figure 5−28. The qualitative match at the saltwater interface was obtained by comparing Figure 7–14b to Figure 7–12.

Some minor adjustments of transport parameters and boundary conditions initially applied in the model were necessary to assure that ammonia concentrations were being calculated properly. In addition, the functional relationship used to represent influx of ammonia from the brine zone was tested with the first-order decay constant assigned to this constituent to assure that the model was capable of reproducing the ammonia concentrations currently observed near the salt-water interface both below the tailings pile and closer to the Colorado River. The ammonia concentrations used as initial conditions in slices 1 and 16 of the model are shown in Figure 7–14.

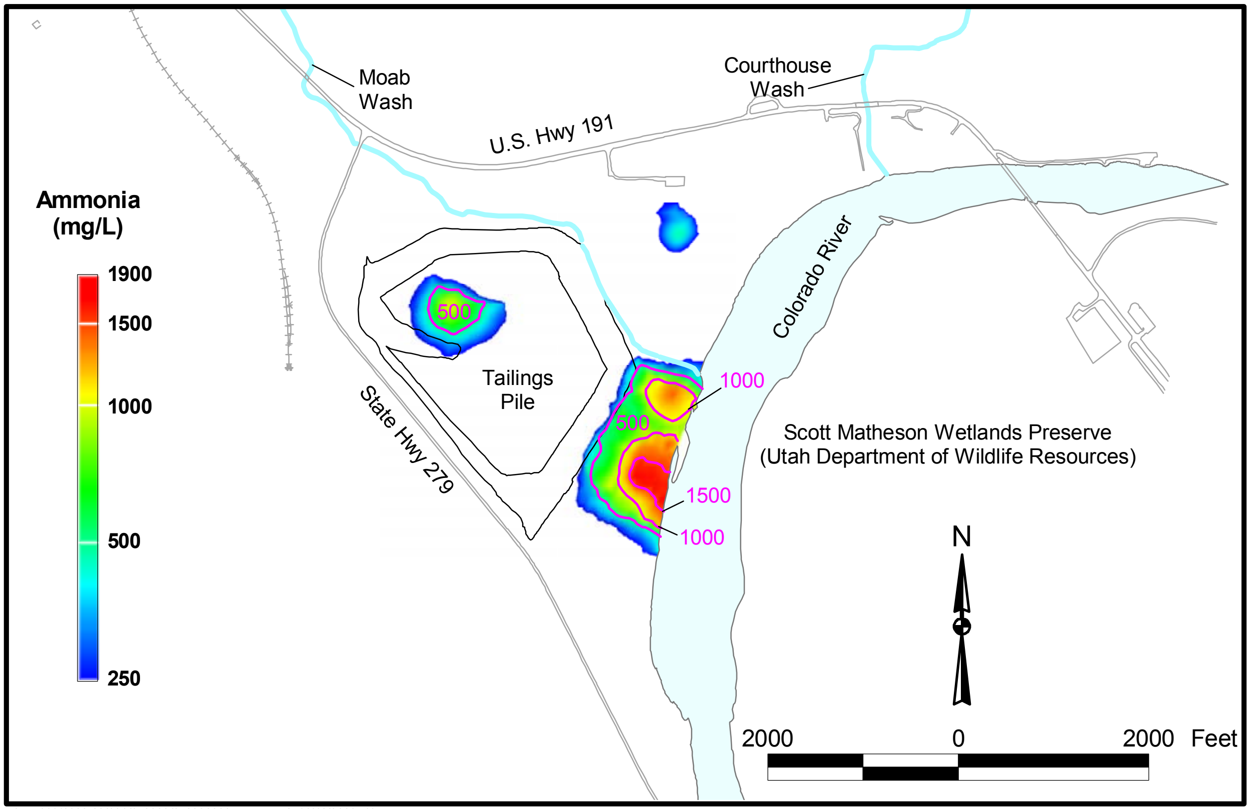

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Figure 7–12. Concentrations of Ammonia at the Saltwater Interface

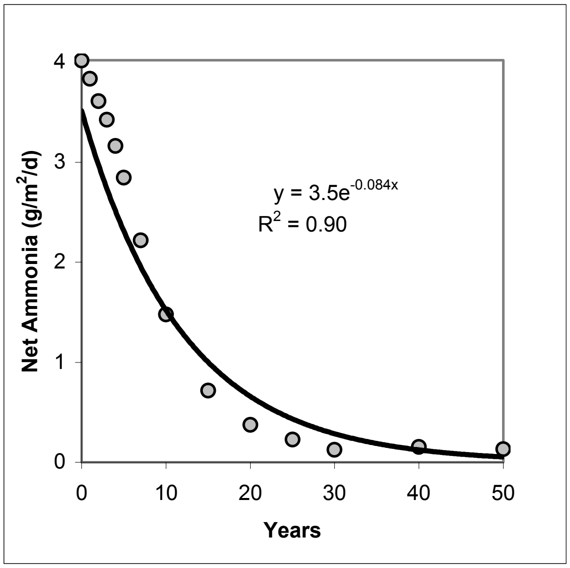
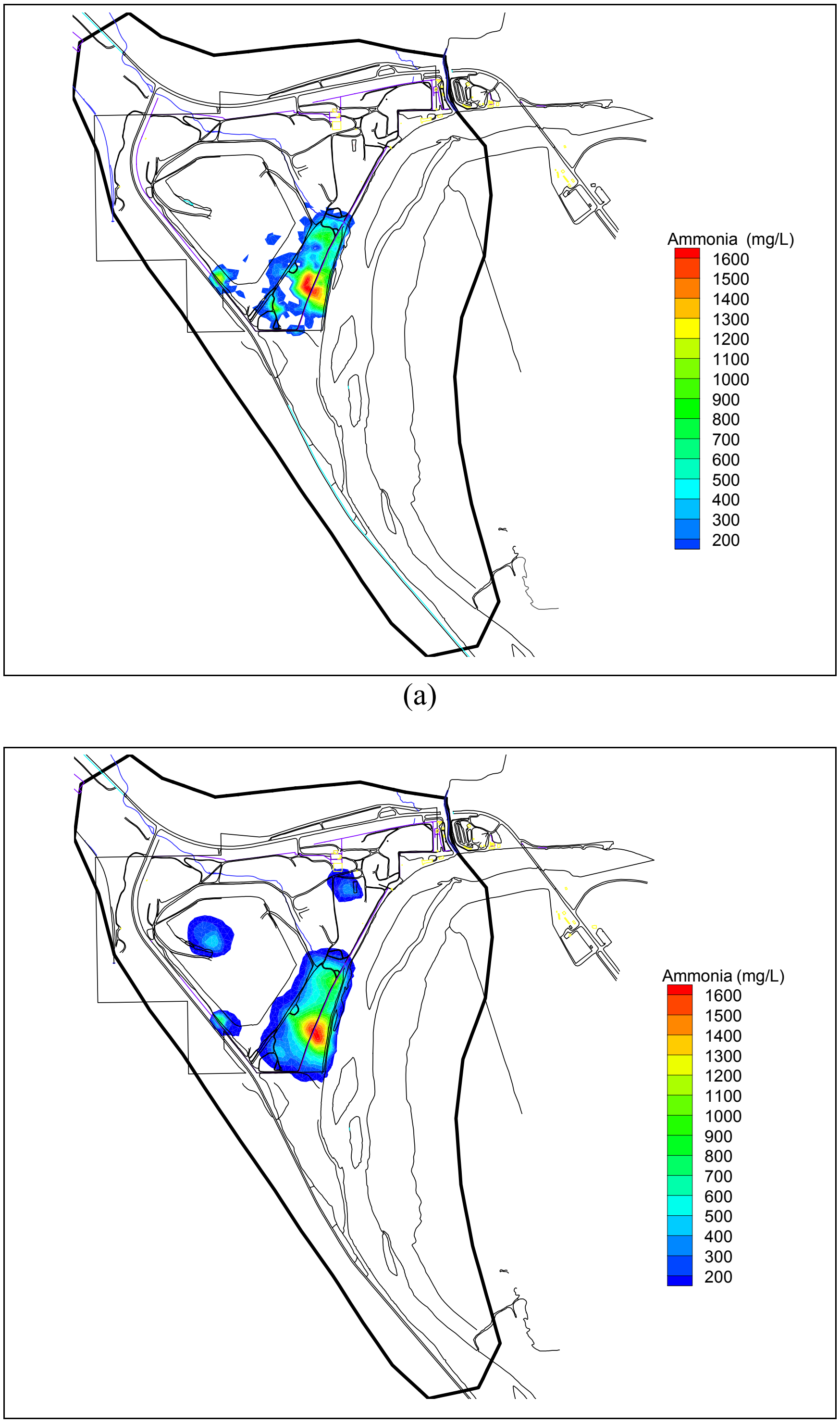


Figure 7–13. Relationship Showing the Ammonia Mass Flux From the Brine Interface to the Overlying Fresher Water System as a Function of Time.



(a)

(b)

Figure 7–14. Initial Concentrations Applied to (a) Slice 1 and (b) Slice 16 of the Moab Flow and Transport Model

Transport parameters applied uniformly throughout the model and concentrations associated with the ammonia source in the tailings pile are summarized in Table 7–6. Also shown is an assumed concentration for ammonia in recharge water entering the freshwater aquifer via bedrock aquifer inflows; this latter parameter was based on measurements of background ammonia concentrations at the site. Along with the calibrated ground water flow model, these transport properties form the "base case" transport model for the site. The base case forms the main predictive tool for the management alternatives considered herein. Variations of the base case look at the predicted effects of the alternative remediation schemes referred to as *off-site stabilization* and *no-action.* These latter remedial alternatives are described in Section 6.9. The base case model also provides the platform upon which model uncertainty analyses were conducted.

Table 7–6. Transport Parameters Input to Base Case Model

Parameter	Value
Effective Porosity	0.3
Longitudinal Dispersivity	25 m
Transverse Dispersivity	2.5 m
Ammonia Retardation (Henry coefficient)	1.325
Ammonia Degradation (1st order constant)	2×10^{-9} sec^{-1}
Background Ammonia Concentration	0.09 mg/L
Ammonia Pore Fluid Concentration	1,100 mg/L

7.5 Predictive Simulations

DOE is currently in the process of preparing an Environmental Impact Statement (EIS) that compares remediation alternatives for cleaning up contaminated ground water at the Moab site. Ground water contaminated with ammonia is discharging to backwater areas along the bank of the Colorado River that may provide suitable habitat for threatened and endangered aquatic species. Resulting ammonia concentrations in these areas are above protective criteria for aquatic species (Section 5.6). Analytical data indicate that ammonia decreases significantly as ground water discharges to and mixes with surface water (a 10-fold decrease is observed on average). A target goal of 3 mg/L ammonia in ground water next to the Colorado River provides reasonable assurance that protective surface water concentrations could be achieved.

Predictions using the ground water flow and contaminant transport model were performed to assess future concentrations of ammonia in the ground water that could be expected as a result of each proposed surface remediation alternative considered in the EIS. Three alternatives are considered in the EIS for remediation of the tailings and other surface source material at the Moab site: (1) no action, (2) on-site disposal, and (3) off-site disposal. The modeling results summarized below were generated using the base case ground water flow and transport parameters modified for each conceptual alternative as described in Section 6.9.

7.5.1 No Action Scenario

Under the no-action scenario, DOE would not perform any short-term or long-term restoration activities or carry out remedial actions to reduce concentrations of site-related constituents in ground water. This alternative would, therefore, rely on natural processes to reduce concentrations of site-related constituents in the ground water. To meet ammonia ground water criteria in ground water adjacent to the river, it would be necessary for the bulk of ammonia capable of discharging to the river to either discharge to the river or attenuate through other processes (e.g., nitrification) during the next 200 years.

One of the natural processes capable of significantly reducing ammonia concentrations in shallow ground water under the no-action alternative is mixing of fresh water entering the system upgradient of the pile with contaminated water entering ground water from tailings seepage. Despite the assignment of a constant ammonia concentration of 1,100 mg/L to tailings fluids (Section 7.4.1.2), the mixing process is expected to achieve greater dilution of ammonia contamination with time as the tailings seepage rate decreases from currently estimated rates. Under the no-action scenario, the tailing seepage rate is predicted to drop from 20 gpm today to a steady-state rate of 8 gpm some 20 years hence (Appendix D, Calculation X0025700), with this latter rate representing a constant source of ammonia for an indeterminate time beyond the 200-year simulation period. This relatively constant source of ammonia for an indeterminate time is what sets the no-action alternative apart from the other two remedial alternatives (see Section 6.0).

Results produced by the base case model under the no-action alternative are shown in Figure 7–15. The concentrations presented in this figure represent the maximum of predicted ammonia-N concentrations at a series of observation locations situated along the west bank of the Colorado River close to the centerline of the plume emanating from the tailings pile. The concentration-versus-time graph indicates that most of the ammonia currently moving through the freshwater flow system and the ammonia re-entering it as influx from the brine zone will naturally flush to the river in approximately 75 years. Predicted concentrations at the river after the 75-year period result from a steady 8 gpm seepage of tailings water containing ammonia at a concentration of 1,100 mg/L. The steady-state ammonia concentration predicted to occur in ground water at the river as a result of this continuing source is approximately 6 mg/L.

It is important to note that the no-action results presented in Figure 7–15 do not account for the eventual dissolution of shallow salt deposits, in the upper 10 ft of the tailings pile, which contain elevated levels of ammonia that equate to concentrations of up to 18,000 mg/L in tailings pore fluids. High ammonia concentrations of this magnitude would likely persist in shallow tailings until such time that the salt deposits are significantly depleted by dissolution via infiltrating precipitation. It is estimated that the solute front associated with dissolution of these salts would take 168 years from present to reach ground water. However, as discussed in Section 6, dispersive transport and sorption processes make it likely that ammonia concentrations in the front will be considerably less than 18,000 mg/L.

In summary, maximum concentrations in ground water near the Colorado River under the no-action alternative are predicted to decline from the current range of 500 to 1,000 mg/L to approximately 6 mg/L about 75 years from now. Thus remediation based on no-action is not expected to meet 3 mg/L target goal. A plan view of predicted ammonia concentrations in ground water 75 years hence under this alternative is presented in Figure 7–16.

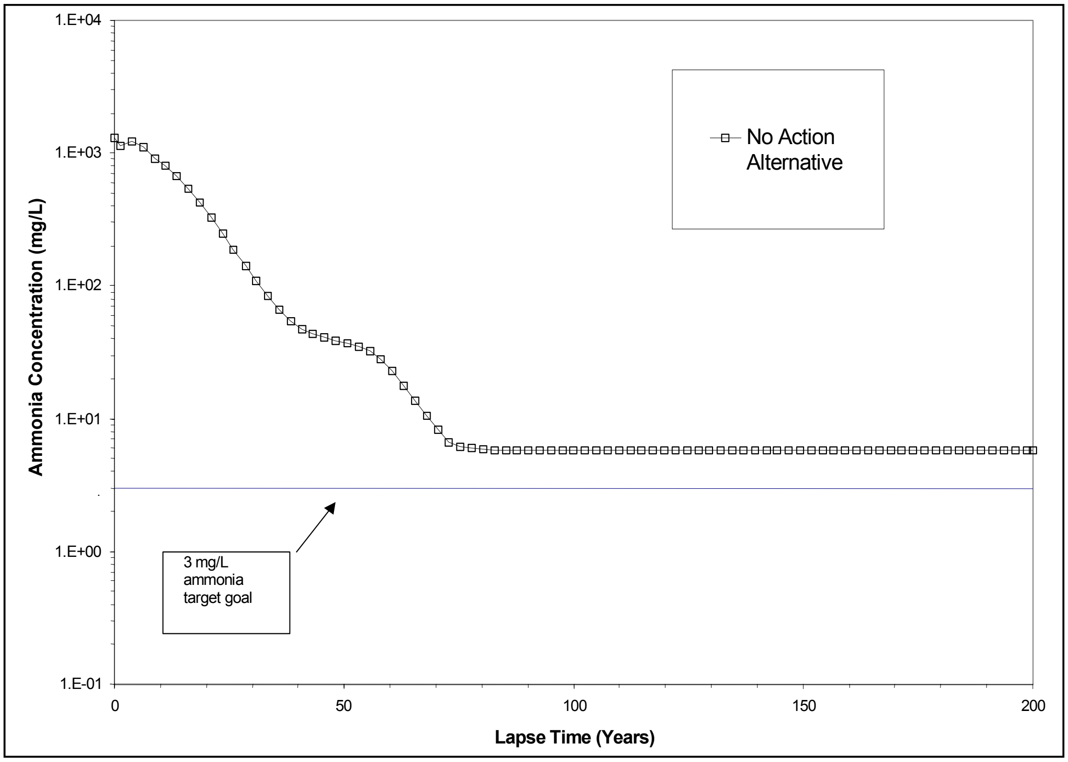


Figure 7–15. Predicted Maximum Ammonia Concentrations in Ground Water Adjacent to the Colorado River for the No-Action Alternative.

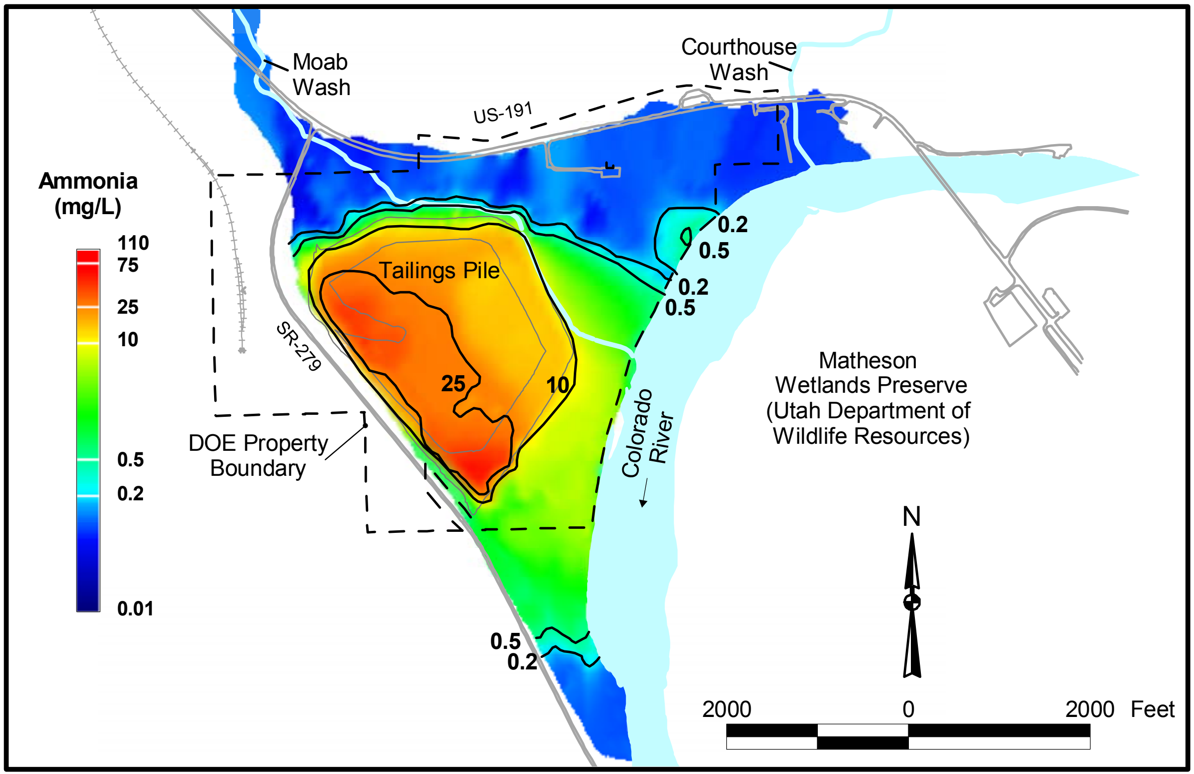


Figure 7–16. Predicted Ammonia (as N) Concentrations in the Ground Water at Steady-State After 75 Years for the No Action Alternative

7.5.2 On Site Disposal Scenario

The on site disposal alternative assumes that the tailings pile will be capped with materials that reduce current tailings infiltration rates by a factor of 10, i.e., to a rate of 1×10^{-8} centimeters per second (cm/s). Thus though this scenario does not completely eliminate the discharge of contaminated water to the Colorado River, it does provide a distinct incremental benefit to affected surface waters as compared to the no-action alternative.

Seepage calculations (Appendix D, Calculation X0025700) indicate that a ten-fold reduction in tailings infiltration would lead to a decrease in the tailings outflow rate from the currently estimated 20 gpm to about 0.8 gpm 122 years hence. Assuming that the ammonia concentration in tailings fluids remains at 1,100 mg/L, the ultimate mass-loading rate of ammonia to groundwater would also be reduced by a factor of 10. The seepage rate from the tailings as a function of time is summarized in Section 6.0.

Because all of the natural processes leading to declines in ammonia concentration in ground water under the no-action alternative are also effective under the on site disposal scenario, predicted ammonia concentrations during the first several decades of simulation would be expected to mimic those shown in Figure 7–15. Significant divergence from the no-action predictions would, however, occur beyond the first 60 to 80 years of simulation.

Predicted maximum ammonia-N concentrations in ground water at the Colorado River over the next 200 years under the on site disposal alternative are plotted in Figure 7–17. As in the case of the no-action alternative, discharge of ammonia now occurring in the freshwater and the brine zones is predicted to be virtually complete after about 70 years. However, instead of leveling off to a constant level at this time, the maximum ammonia concentration continues to decrease and reaches the 3 mg/L criterion in about 80 years. Subsequent decreases in ammonia levels eventually lead to a constant maximum ammonia concentration of about 0.7 mg/L in approximately 170 years.

Like the results from the no-action analysis, the concentration-versus-time predictions shown in Figure 7–17 do not account for dissolution of high levels of ammonia observed in shallow salt deposits in tailings. It is estimated that, with a tailings cover design that limits infiltration to 1×10^{-8} cm/s, the relatively high ammonia concentrations resulting from dissolution of these salts would reach the ground water 1,094 years from the present. The potential for significantly higher ammonia concentrations associated with the salt deposits to enter the ground water is discussed in Section 6. In all likelihood, attenuation processes will cause ammonia levels in tailings fluids near the base of the pile to be considerably less than the 18,000 mg/L that currently results from salt dissolution in the shallow portion of the pile.

To summarize, ammonia concentrations in ground water near the river would be expected to decline under the on site disposal scenario from the currently observed range of 500 to 1,000 mg/L to the target goal of approximately 3 mg/L in 80 years, and to about 0.7 mg/L when steady-state concentrations are reached in about 170 years. Map views of predicted ammonia concentrations in shallow ground water after 80 years and 200 years of simulation with this alternative are presented in Figure 7–18 and Figure 7–19, respectively.

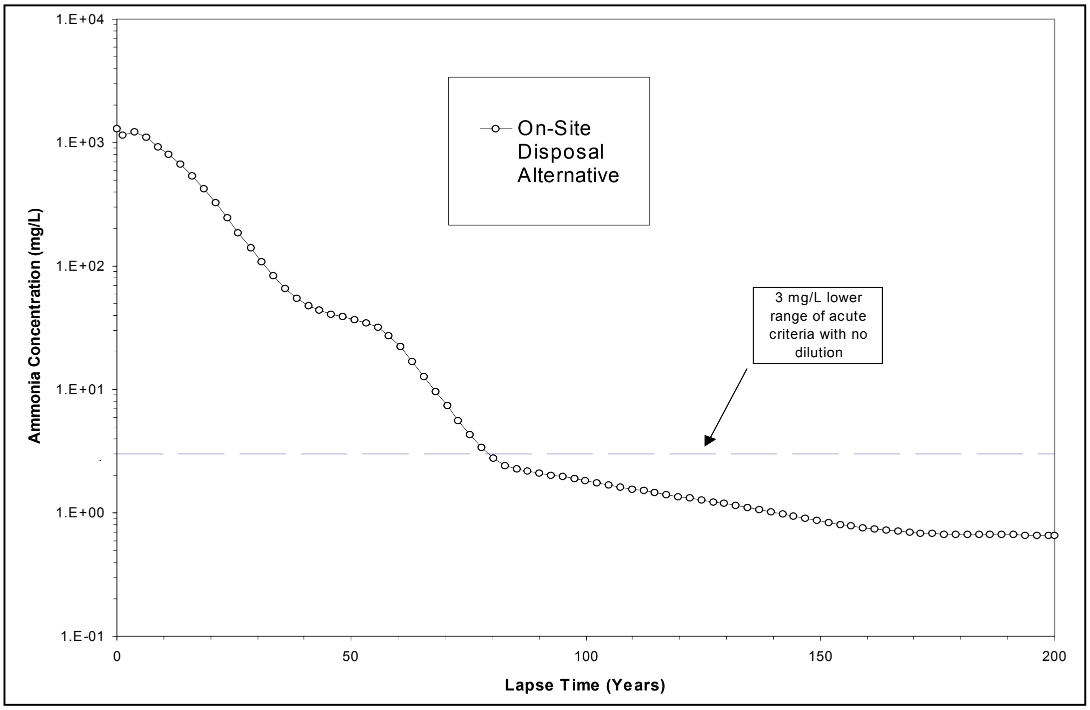


Figure 7–17. Predicted Maximum Ammonia Concentrations in Ground Water Adjacent to the Colorado River for the On-Site Disposal Alternative

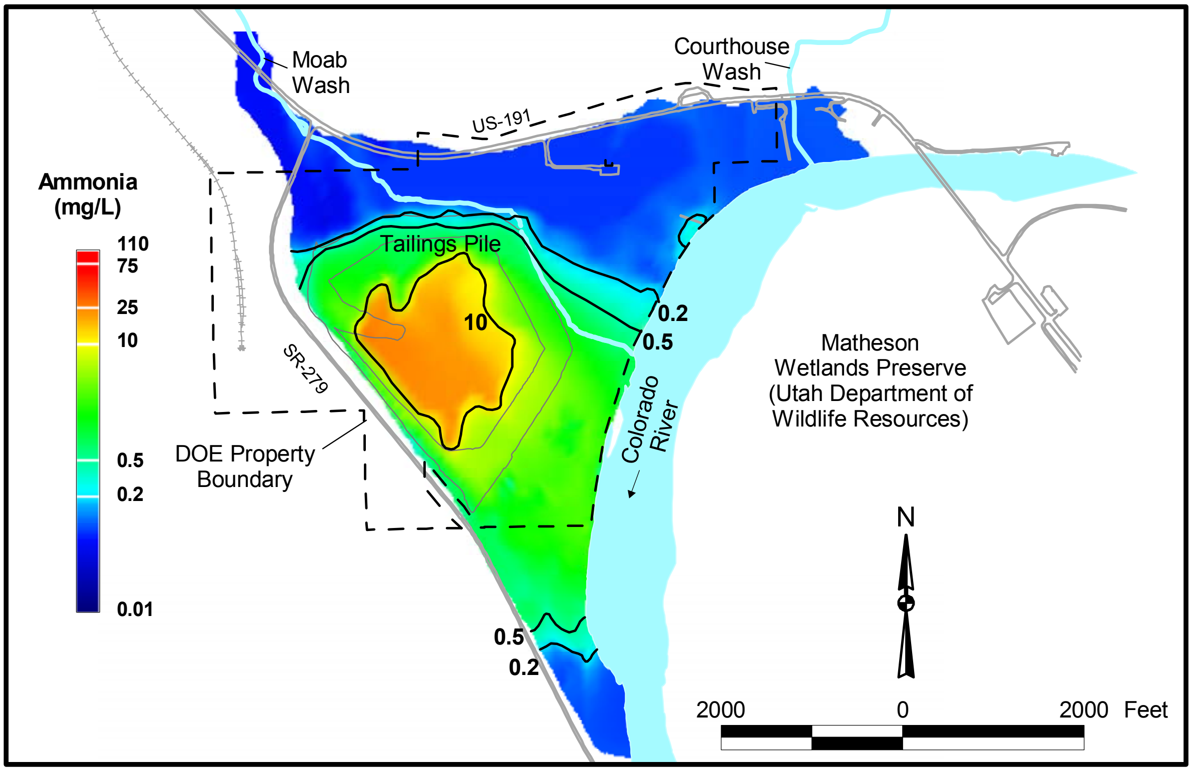


Figure 7–18. Predicted Ammonia Concentrations in the Ground Water After 80 Years for the On-Site Disposal Alternative.

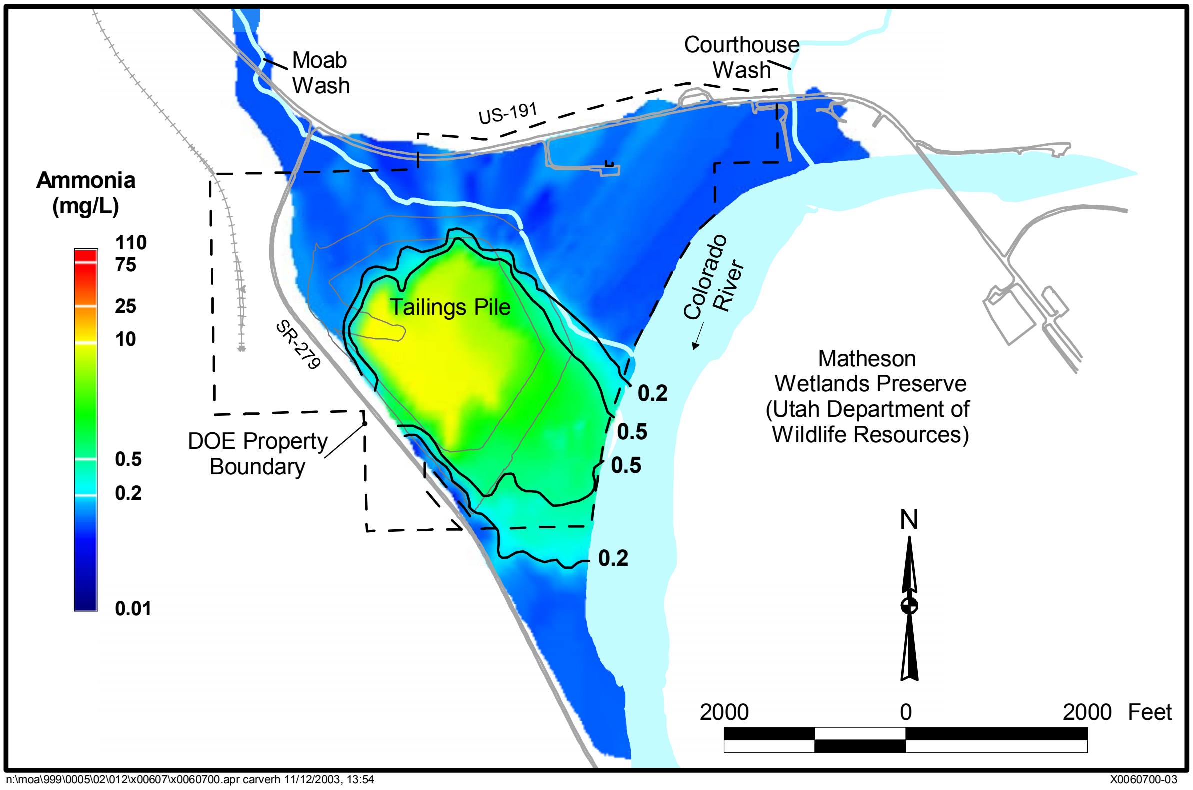


Figure 7–19. Predicted Ammonia Concentrations in the Ground Water at Steady-State After 200 Years for the On-Site Disposal Alternative.

7.5.3 Off Site Disposal Scenario

Removing the tailings pile from the Moab site would reduce concentrations of site-related constituents in ground water and potentially lead to ammonia discharges to the river that are no greater than those that occur under ambient, background conditions. To assess the effects of this off-site disposal alternative on future ammonia concentrations in ground water near the river, it was necessary to account for the years that would be required to carry out all associated remediation steps. This was accomplished by assuming that the tailings would be completely removed from the site by the end of the year 2010. Until this time, seepage rates from the tailings were assumed to decrease gradually from the estimated current rate of 20 gpm to a lower value predicted in the previously mentioned calculation (Appendix D, Calculation X0025700) dealing with this process; the associated ammonia concentration in tailings fluids during this time period was again set at 1,100 mg/L. After 2010, a steady infiltration rate of 8 gpm was applied, which is equivalent to the average volumetric rate of recharge that would occur over the tailings area under the assumption that 10 percent of annual precipitation naturally recharges the aquifer. It was assumed that there would be no dissolved ammonia in this natural recharge.

Maximum ammonia-N concentrations in ground water at the river and downgradient of the tailings pile, as predicted by the 3-D model for the off site disposal alternative, are presented in Figure 7–20. These results are similar to those previously shown for the no-action and on site disposal alternatives in that most of the ammonia currently observed in the freshwater and brine zones is predicted to flush to the river within a period of about 75 years. After this initial period, however, the maximum ammonia concentration drops steadily for the next 70 years and eventually reaches a constant concentration of less than 0.01 mg/L. This latter ammonia level results from a mixing of bedrock aquifer water discharging to the ground water system at a concentration of 0.09 mg/L with water containing no dissolved ammonia. Sources of water containing no ammonia include the subsurface flow through Moab Wash and areal recharge from precipitation.

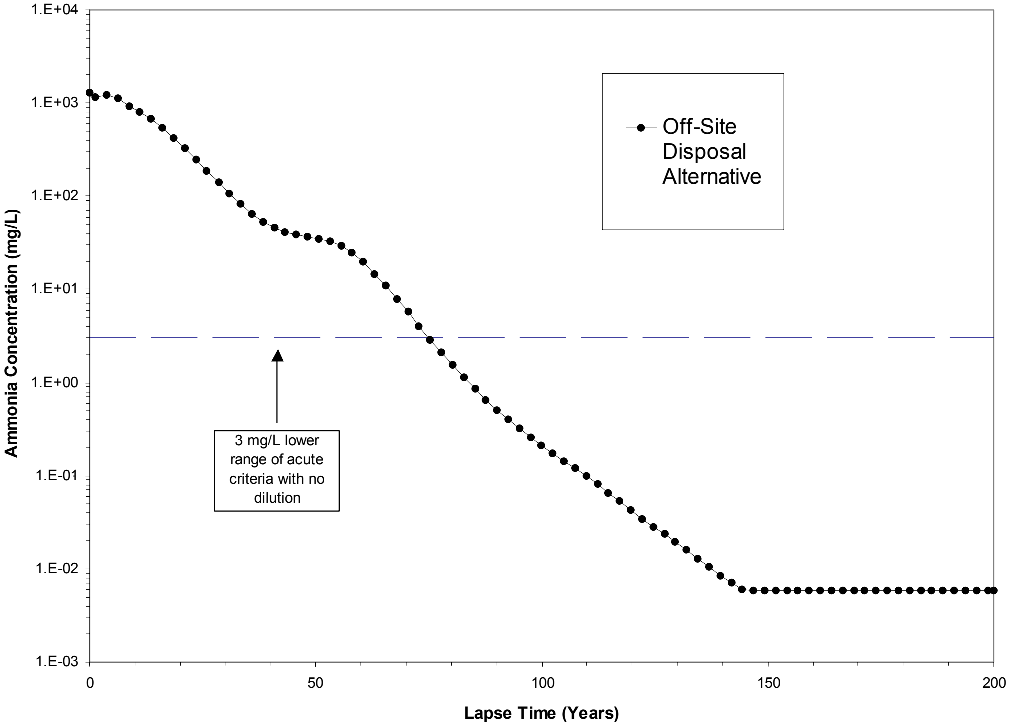


Figure 7–20. Predicated Ammonia Concentrations in Ground Water Adjacent to the Colorado River Under the Off-Site Disposal Alternative

Modeling results for the off site disposal alternative indicate that ammonia concentrations in ground water near the bank of the Colorado River would be expected to decline from the current 500 to 1,000 mg/L to the target goal of approximately 3 mg/L in 75 years and to reach background concentrations in approximately 145 years when steady-state conditions are achieved. A map view of predicted ammonia concentrations in shallow ground water at 75 years hence under this alternative is provided in Figure 7–21. These outcomes imply that that the off-site disposal alternative stands the greatest chance of reducing ammonia levels in the Colorado River to those that would be observed under ambient conditions.

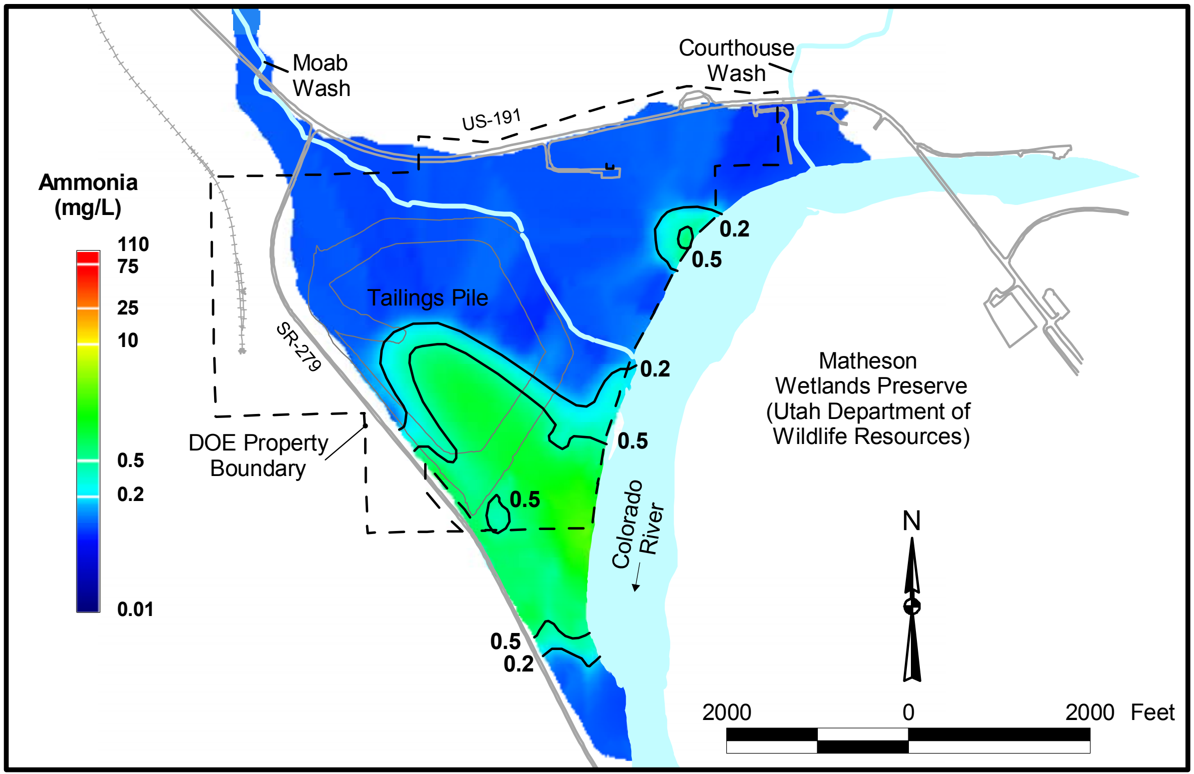


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Figure 7–21. Predicted Ammonia Concentrations in Alluvial Ground Water After 75 Years Under the Off-Site Disposal Alternative

7.5.4 Tailings Inundation and 1984 Flood

The potential effects of a significant ground-water rise caused by flooding in the Colorado River were also examined with the 3-D flow and transport model. Ground water levels observed during a major flood in 1984 were approximated in the model to assess how saturation of the basal portion of the tailings would impact ammonia discharges to the Colorado River. Because this simulation assumed that the tailings pile would still be present, it was considered applicable to the no-action and on-site disposal alternatives. The simulation was somewhat simplified in that it predicted ammonia concentrations resulting only from drainage of tailings fluid; i.e., ammonia concentrations stemming from migration of the existing ammonia plume in ground water was not evaluated.

During the 1984 flood, river discharge reached a peak of 70,300 cfs (Calculation X0060200). A water level hydrograph for well ATP-3 at the time showed that ground water near the northwestern part of the pile reached a maximum elevation of 3,966 ft above msl. For comparison, boreholes drilled during site characterization indicate that the base of the tailings along the pile's southeast margin is at an elevation of about 3,963 ft above msl. Consequently, some saturation of the basal part of the tailings likely occurred in 1984.

A conservative estimate of the volume of tailings that became saturated during the 1984 flood event was derived by assuming that ground water reached a uniform peak height of 3,966 ft above msl near the southeast margin of the tailings. Taking into account the gradual rise in elevation of the tailings base with distance to the northwest, the saturated volume was calculated to be 2,365,000 cubic feet (ft^3). Under the assumption that the affected tailings had a drainable porosity of 25 percent, the total volume of water available for relatively quick drainage from the pile after passage of the flood was in turn calculated to be 591,250 ft^3, or 4,422,550 gallons. By further assuming that this volume of water drained from the pile at a constant rate for ten consecutive days, the rate of seepage back into the ground water system was calculated to be 307 gpm. Assuming that the ammonia concentration in this draining water was 1,100 mg/L, the mass loading rate of ammonia to ground water was calculated to be approximately 1.84×10^6g/d. These rates were applied uniformly in the flow and transport model over a zone that approximated the plan view area affected by the flood-induced saturation.

Model results from the flooding scenario in terms of maximum predicted ammonia concentrations adjacent to the river are presented in Figure 7–22. This concentration-versus-time graph suggests that it would take about ten years for peak ammonia levels stemming from flood effects to reach the river, and that ammonia concentrations adjacent to the river at that time would slightly exceed 2 mg/L. The simulation results also suggest that a subsequent time span of about 20 years would be required to remove all residual ammonia associated with the flood effects.

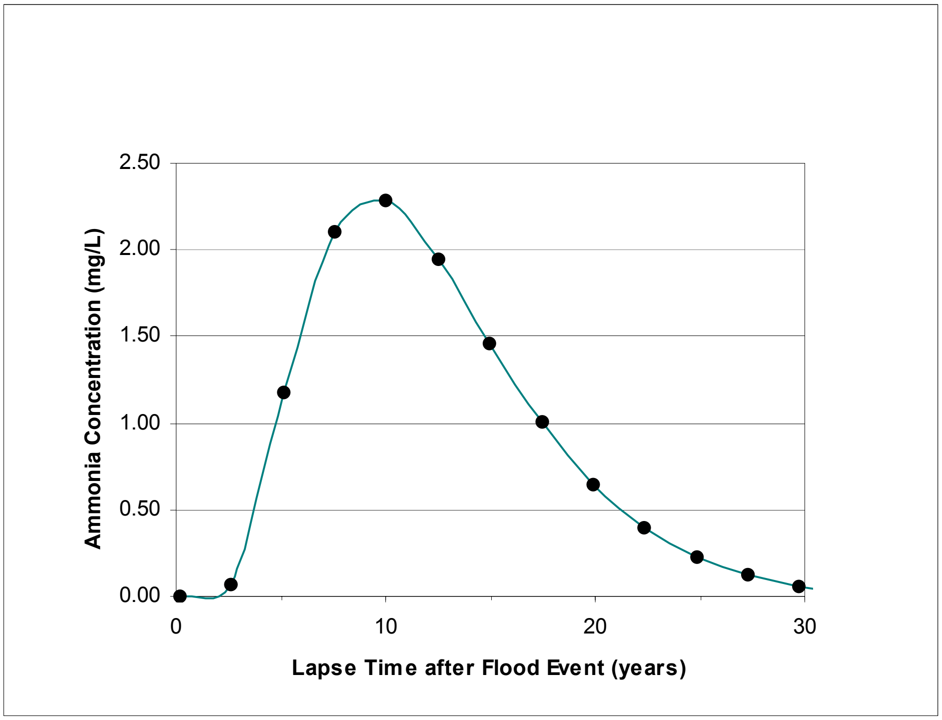


Figure 7–22. Maximum Ammonia Concentration in Ground Water Near the Bank of the Colorado River Predicted After a 70,300 cfs Flood Inundates the Base of the Tailings Pile

7.6 Prediction Uncertainty Analysis

Several sources of uncertainty affect the predictive modeling described in Section 7.5, with much of it being associated with the flow and transport parameters used in the model. Model predictions based on highly uncertain parameters could also be highly uncertain, which could lead to poor management decisions. To assess the potential effects of parameter uncertainty on simulation results, additional flow and transport model runs were made using parameter values that were perturbed from those applied in the base case model. All of the additional simulations were based on the on site disposal alternative.

In keeping with the focus of the predictive simulations discussed in Section 7.5, predicted ammonia concentrations in ground water adjacent to the Colorado River were used to gage the effects of parameter uncertainty on model uncertainty. This performance measure was represented by the average ammonia concentration computed at eight monitor well locations (MOA-401, -402, - 403, -405, -406, -407, -409, and SMI-PZ1S) 80 years in the future. The eight monitor wells were located between the southeast toe of the tailings and the river.

7.6.1 Flow Parameter Uncertainty

Five ground water flow model parameters were included in the predictive model uncertainty analysis (Table 7–7). Minimum and maximum values assigned to each were partly based on results of the flow model sensitivity analysis discussed in Section 7.3.5.5. A criterion placed on the selection was that the parameter value used had to result in an acceptable flow model calibration. As discussed in Section 7.3.5.3, model calibration was considered acceptable if the ratio of residual standard deviations to the range in observed hydraulic heads had a value of 0.1 or less.

Table 7–7. Summary of Flow Parameters Used in Prediction Uncertainty Analysis

Parameter	Min	Base Case[a]	Max	Note
Hydraulic Conductivity (ft/day)	80	147	177	Min and max based on sensitivity analysis
Glen Canyon Flux (gpm)	50	131	200	Min and max based on sensitivity analysis
Moab Fault Flux (gpm)	200	254	420	Min and max based on sensitivity analysis
Evapotranspiration (gpm)	30	147	200	Min and max based on sensitivity analysis
Recharge (m/yr)	0.004	0.02	0.04	2%, 10%, 20% of annual precip., respectively

[a] Value used in base case calibrated model

The predictive simulations made with the minimum and maximum parameter values listed in Table 7–7 indicated that only three of them – gravel hydraulic conductivity, Moab Fault inflow, and volumetric ET rate – showed a significant effect on ammonia concentrations near the Colorado River. Results of the prediction uncertainty analysis for these flow parameters, are shown in the bar graphs presented in Figure 7–23. Ammonia concentrations computed by the model using the minimum and maximum input values are indicated along with the comparable concentration produced by the base case model. For all three of the flow model parameters considered, the maximum predicted ammonia concentration is virtually identical to the concentration predicted with the base case model. In contrast, the predicted minimum ammonia concentrations for all three perturbed parameters are about half of the concentration computed by the base case model. This decrease in concentrations suggests that it might be possible for ammonia levels in ground water near the river to eventually reach a concentration less than the 3 mg/L criterion under the no-action alternative.

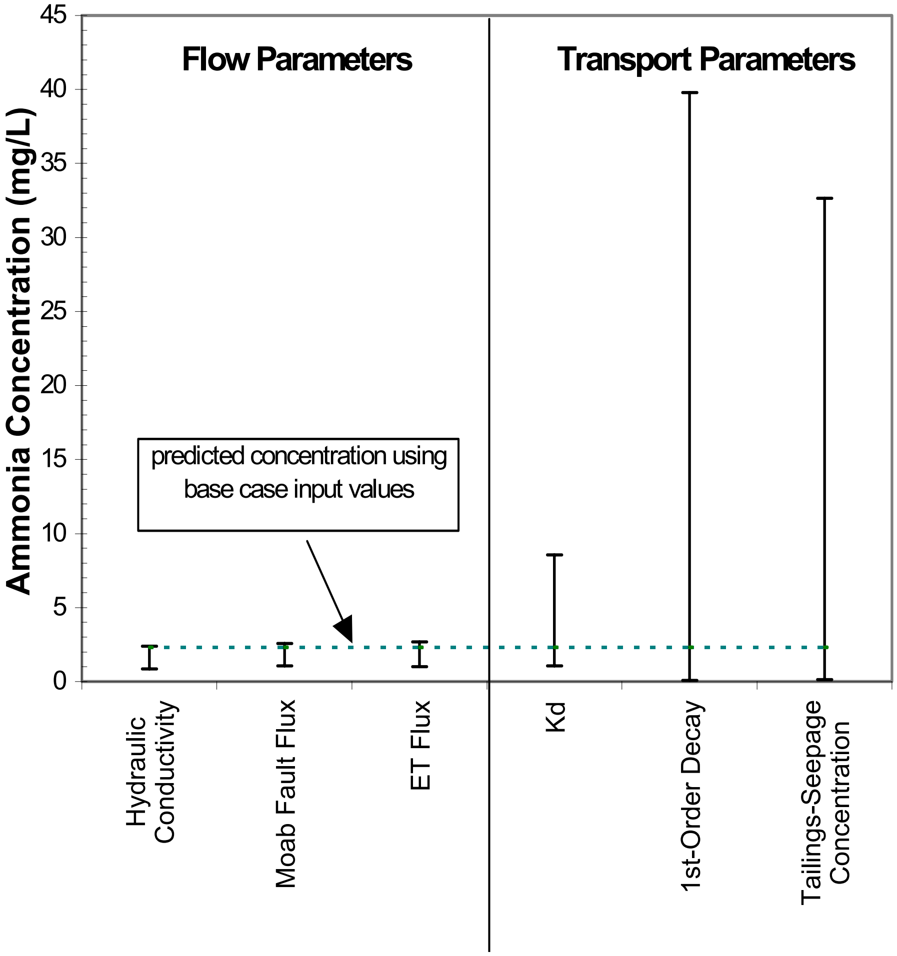


Figure 7–23. Predicted Maximum and Minimum Ammonia Concentrations in Ground Water Near the River at 80 Years Resulting from Uncertainty Analysis for the On-Site Disposal Alternative

7.6.2 Transport Parameter Uncertainty

Five transport parameters, listed in Table 7–8, were included in the predictive model uncertainty analysis. Minimum, maximum, and base case input values for these are listed in Table 7–8 along with notes describing the methods used to select the values.

In general, transport parameters included in the uncertainty analysis were found to have a much greater impact on predicted ammonia concentrations than did flow model parameters. Of the five parameters considered, the three that were chemical-specific (ammonia distribution coefficient [K_d], ammonia degradation rate constant, and ammonia concentration in tailings fluids) showed the greatest impacts. The results of perturbing each of these latter parameters around base case input values are shown in Figure 7–23.

The predictions stemming from uncertainty in transport parameters indicate that ammonia concentrations in ground water near the river under on site disposal alternative could exceed the target concentration of 3 mg/L by more than an order-of-magnitude.

Table 7–8. Summary of Transport Parameters Used in Prediction Uncertainty Analysis

Parameter	Min	Base Case[a]	Max	Note
Dispersivity (m)				
Longitudinal	2.5	25	50	1%, 5% and 10% model scale, respectively (Gelhar et al. 1992)
Transverse	0.25	2.5	5	10% of long. dispersivity (Gelhar et al. 1992)
Ammonia Distribution Coefficient (mL/g)	0	0.5	3	Based on calculation Moab 12-2002-03-05-00
Effective Porosity (alluvium %)	0.2	0.3	0.4	Morris and Johnson 1967
1st Order Decay Rate (1/sec)	2E−10	2E−09	2E−08	Based on literature values (base case, Kipp 1987)
Ammonia Tailings Seepage (mg/L)	100	1,100	18,000	Based on Calculation X0025700

[a] Value used in base case calibrated model

7.7 Quality Control

Quality control for this modeling project was implemented during all phases of the work. Development of the site conceptual model required review of previous published and unpublished reports, surface geological mapping, lithologic logs, results of aquifer tests, results of ground water sampling, and laboratory tests. These sources of information were honored during the development of the numerical flow and transport model and in assigning model parameters and boundary conditions. ASTM guidelines were used to apply the numerical ground water model to site conditions, to calibrate the model, to perform sensitivity and uncertainty analysis, and to document the modeling report.

Quality control measures were implemented to enhance model credibility and to facilitate independent, third party review of the modeling product. Records created during the modeling process serve to track the decisions and operations that were required to complete the project. These records include summaries of individual simulations, copies of important input files and Fortran programs, records of pivotal conversations and correspondence, verification of model input parameters and boundary condition, and data archival. This section describes the links that exist between the numerical model and the supporting information. References throughout this section direct the reader to Attachment 1, which is contained in electronic format on CD. The information was written to CD to trim the physical size of the Appendix while bolstering the information provided to the interested reader.

7.7.1 Input Files and Fortran Programs

Versions 4.8 and 5.0 of FEFLOW were both used during the simulations described here. Results from the two versions are not always in complete agreement because double precision is used in FEFLOW 5.0 and only single precision in FEFLOW 4.8. Comparisons indicate that identical results occur when identical input files are passed to different machines running the same version of FEFLOW.

Attachment 1 contains three input files: moabmxk311.fem, moabmxk312.fem, and moabmxk313.fem. These are the FEFLOW 5.0 input files for the on-site disposal, off-site disposal, and no-action alternatives, respectively. Investigators wishing to use earlier versions of FEFLOW to evaluate these files should contact the developers (Stoller) for files saved to an earlier version of FEFLOW.

7.7.2 Significant Correspondence

Attachment 1 contains records of significant email correspondence between Stoller and WASY Software Development. The file "email-01" documents the correspondence that occurred regarding topics (A) and (B) below. The file "email-02" documents the correspondence that occurred concerning topic (C) below:

(A) Convective versus Divergence Solver

An option exists in the Options menu of the FEFLOW preprocessor where the user may select the convection or divergence form of the solver. Results from the mass transport modeling are particularly sensitive to the choice of this option particularly when using 2nd-kind mass flux boundary conditions, and the results differ depending upon whether the boundary condition is a source or a sink. It was found that nodal water balances calculated on source nodes yield correct concentrations and mass fluxes if the divergence form is selected. If the convective option is selected, the mass accumulates in the system above specified values. Because it yields accurate nodal mass fluxes for the 2nd-kind mass transport boundary conditions, the divergence form of the solver was used for source nodes, namely the tailings pile.

(B) 2nd-Kind Mass-Transport Boundary Conditions for Sink Nodes

Initially, it was intended to use the 2nd kind boundary condition for mass at the sink nodes because this capability would have enabled the evaluation of variable ammonia-mass reduction rates for the tamarisk. However, it was found that if the 2nd kind boundary condition on sink nodes were used with the divergence form of the solver, the resulting nodal concentrations were 20 to 30 percent less than the quotient obtained when dividing the nodal mass flux by the nodal liquid flux. Based on this finding it was concluded that the most consistent results would be obtained by using the 2nd-kind boundary condition only for liquid fluxes in the tamarisk. The mass removed by the tamarisk would therefore equal the nodal concentration divided by the nodal liquid flux.

(C) Kd/Henry Sorptivity Coefficient Calculation

This correspondence occurred to verify that Kd values were being assigned correctly. The FEFLOW model uses a term, designated the *Henry Sorptivity Coefficient*, to account for the partitioning of the solute on the liquid and solid phase. The second email provides confirmation that the Henry Sorptivity coefficient was being applied correctly.

7.7.3 Summary of FEFLOW Simulations

The flow portion of the model required approximately 300 simulations in order to achieve calibration. After this, the model was run approximately 80 additional times to evaluate management alternatives and to study parameter sensitivity. A listing of each model scenario that was evaluated after model calibration was achieved is contained in Attachment 1.

7.7.4 Simulation Logs

A simulation log was created to document the results of each model scenario and each parameter-sensitivity evaluation. Each simulation log provides a summary of parameters that were used to create an evaluation. Copies of the simulation logs for each of the simulations beginning with moabmxk311.fem (the basecase simulation) are contained in Attachment 1.

7.7.5 QA/QC Checklist

A listing of the quality assurance/quality control (QA/QC) checklist that was prepared to document that each parameter and boundary condition for the model was checked and verified is contained in Attachment 1. Discrepancies discovered as part of the QA/QC process were noted on the checklist.

The most significant discrepancy that surfaced during the QA/QC checks was that FEFLOW 5.0 yields dry cells in 3 cells in Layer 1 due to the low hydraulic conductivity in those cells combined with the specified evapotranspiration flux. The dry cells do not occur in FEFLOW 4.8. This result shows once again that with regard to the flow model, evapotranspiration is a sensitive parameter. Dry cells in Layer 1 yield negative concentrations in the flow field at the affected nodes. However, there is no difference in the model outcome after steady state conditions become established.

7.7.6 Data Archival

Simulations completed as part of this project are archived on the computer network at the U.S. Department of Energy office in Grand Junction, Colorado. All the FEFLOW input files are archived on network drive Vol 4 on 459\HOME\M50122\Moabmodl\FEFLOW. Any of these files can be obtained by placing a request to the developers (Stoller).

7.8 Summary and Conclusions

A model was developed to simulate 3-D ground water flow and transport in the freshwater zone portion of the alluvial aquifer underlying the Moab site. The model was designed to account for contributions of contaminants to the freshwater zone from both the tailings pile and a deeper brine zone. Key hydrogeologic features of the Moab site, such as the Colorado River, were incorporated in the model, making it possible to replicate currently observed ground water flow processes and patterns.

Predictive ammonia transport simulations were performed for the three surface remediation alternatives considered in the EIS. The forecasting horizon for the predictions was 200 years. A simulation was also performed to evaluate the effects of re-saturating basal portion of the tailings pile in the event of a major flood on the Colorado River.

Brief qualitative and quantitative summaries of the ground water flow and transport model are presented in following sections. These are followed by a final discussion of model predictions, including those resulting from an assessment of the effects of model parameter uncertainties.

7.8.1 Qualitative Summary

The ground water flow portion of the model was calibrated using observed water levels at 38 on-site monitor wells. Steady-state conditions were simulated under the assumption that the local ground water system exists in a state of dynamic equilibrium.

Steady-state ground water levels produced by the calibrated model (Figure 7–9) closely resemble observed ground water elevations in the alluvial aquifer (Figure 5−13). The results of particle tracking calculations conducted in the shallowest layer of the model, as presented in Figure 7–24, show that ground water flows mostly southeastward at the site. Most of this ground water enters the alluvial aquifer flow system as discharge from the Glen Canyon and Entrada Sandstone bedrock aquifers. The particle tracks in Figure 7–24 indicate a slight amount of divergent flow as ground water coming from these sources underflows the tailings pile and discharges to the tamarisk area and to the Colorado River. This flow pattern is consistent with the site conceptual model.

The transport portion of the 3-D model was not calibrated quantitatively. However, several test simulations were performed with the transport model to assure that it was accurately computing dissolved ammonia concentrations. In addition, a test of ammonia influxes from the brine zone produced an ammonia plume in ground water (Figure 7–14) that was similar in shape, concentration, and location to observed (Figures 5−28 through 5−30).

7.8.2 Quantitative Summary

Table 7–9 presents a comparison of model-computed water budget components with those estimated based on the site's hydrogeologic characteristics (Section 5.2.3 and Table 5−5). As this table indicates, simulated inflows and outflows compare favorably with their estimated equivalents. Moreover, the model-computed total flow through the site ground water system falls within the range of estimated total system flows.

The metric used to evaluate the calibration performance for the ground water flow model consisted of the ratio of the standard deviation of head residuals to the range in observed heads at the site. The goal of the calibration effort was to achieve a value for this ratio that was less than 0.1 (i.e., 10 percent). During calibration, aquifer hydraulic conductivities and some boundary conditions were adjusted such that a value of 0.055 for the metric was ultimately achieved. Analysis of the head residuals resulting from the calibration demonstrated that the flow model has a slight negative bias, i.e., the model tends to compute hydraulic heads that are slightly larger than observed heads. A scatter plot of computed heads versus observed heads was also used to evaluate calibration performance. Most values posted on the graph fell on a straight line and a linear regression analysis of computed and observed heads produced a relatively large coefficient of determination ($R^2 = 0.93$); both of these results were indicative of a successful calibration.
A sensitivity analysis conducted with the calibrated flow model indicated that computed hydraulic heads are most sensitive to bedrock aquifer inflow along the Moab Fault, ET flux from tamarisk, and hydraulic conductivity of the gravels comprising most of the alluvial aquifer.

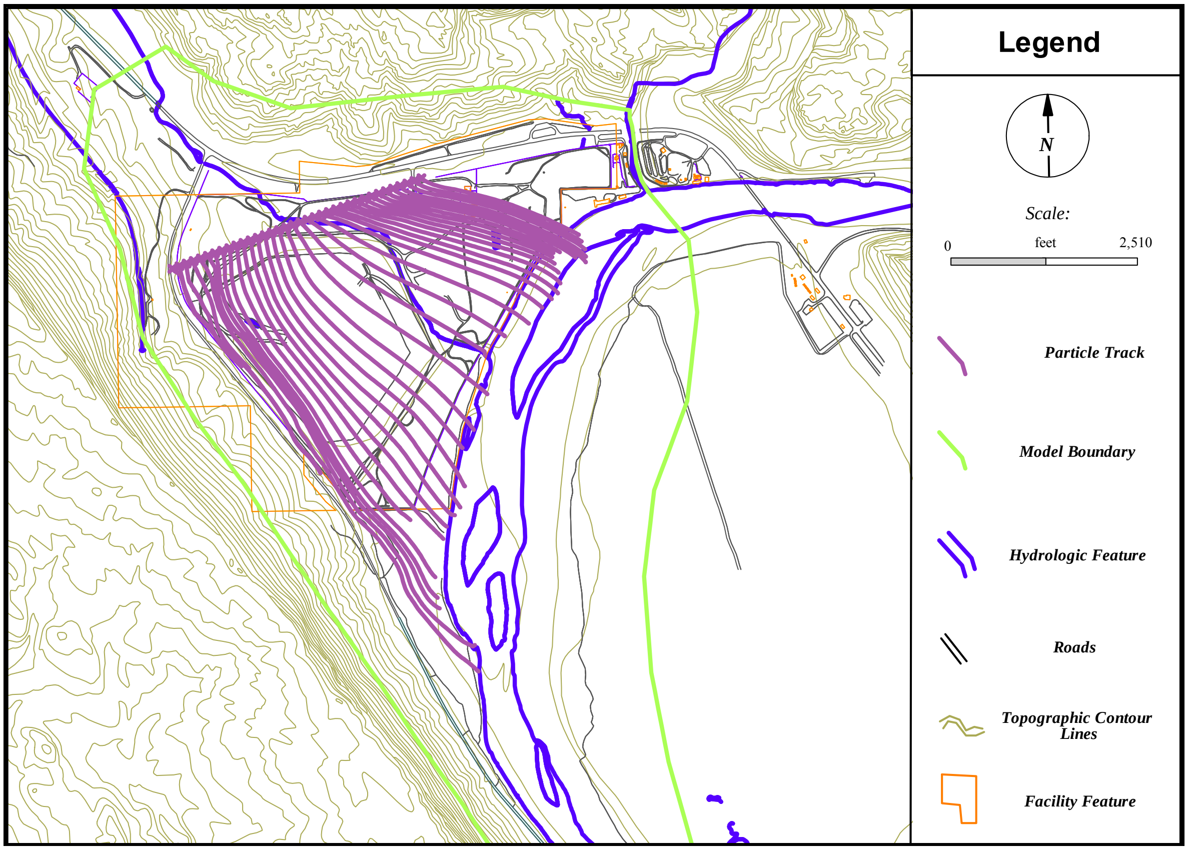


Figure 7–24. Particle Tracking Results for Layer 1 of Calibrated Flow Model

Table 7–9. Comparison Between Estimated Water Budget and Model-Simulated Water Budget

Flow Component	**Estimated**		**Model Calculated**	
	Inflow (gpm)	**Outflow (gpm)**	**Inflow (gpm)**	**Outflow (gpm)**
Areal Recharge	16.3−65	0	26.3	0
Moab Wash	0.5−33	0	0.9	0.3
Glen Canyon Group and Moab Fault	28−280	0	374	0
Tailings Pile	20	0	17	
Evapotranspiration		208−504		148.5
Colorado River		300−460	4.6	274.5
Total	65−400	500−960	422.7	423

Estimated inflow and outflow represents a best estimate based on the range of values appearing in Table 5−5.

7.8.3 Model Predictions and Uncertainties

Predictive runs with the 3-D model focused on computed ammonia concentrations in ground water adjacent to the Colorado River over the next 200 years under each of three remedial alternatives. Each alternative was evaluated for its ability to ultimately meet a near-river target ammonia concentration of 3 mg/L, a value considered protective of aquatic species in surface water near the river’s west bank.

The predictive simulation of future conditions under the no-action remedial alternative indicated that the maximum ammonia concentration in ground water adjacent to the Colorado River decline from the current 500 to 1000 mg/L to about 6 mg/L in about 75 years hence. Thereafter, the near-river ammonia concentration would maintain a value of 6 mg/L for an indeterminate time, suggesting that this alternative would not be protective of aquatic species in the river.

Under the remedial alternative referred to as on site disposal, maximum ammonia concentrations in ground water adjacent to the river were predicted to drop to 3 mg/L in about 80 years. Subsequently, near-river ammonia concentrations continued to decrease to a value of about 0.7 mg/L after 200 years. The model predicted that this latter concentration would be maintained for an indeterminate time due to continued loading of ammonia from tailings fluids, albeit at a slower rate than that occurring under the no-action alternative.

Simulation of the off site disposal scenario indicated that maximum ammonia concentrations in ground water adjacent to the river would reach a background concentration of less than 0.01 mg/L about 150 years from now. Similar to the results from the on site disposal simulation, the model predicts that near-river ammonia concentrations would reach the protective criterion of 3 mg/L in about 75 years.

An uncertainty analysis conducted with the transport model under the on site disposal alternative indicated that ground water flow parameters have relatively mild impacts on computed ammonia concentrations, and that chemical-specific transport parameters have the greatest impacts. The uncertainty analysis revealed that certain values for ammonia degradation rate constant and the concentration of ammonia in tailings fluids were capable of producing ammonia concentrations in ground water near the Colorado River that were at least an order of magnitude larger than those predicted with the calibrated model. On the other hand, the uncertainty analysis suggested that feasible, relatively large values of the ammonia-degradation constant and reasonably small values of the tailings fluid ammonia concentration could lead to insignificant ammonia concentrations near the river under the on site disposal alternative. These latter observations might also apply to the no-action remedial scenario.

Encroachment of ground water into the base of the tailings during a large runoff event on the Colorado River was also evaluated with the model. Conditions observed during a 1984 flood were approximated in the model to predict ammonia concentrations in ground water near the river after the flood wave passed. In lieu of accounting for transport of ammonia currently observed in site ground water, this simulation assumed that the only contributions of ammonia to ground water would occur as a result of drainage from saturated tailings affected by the flood. Model predictions suggest that near the Colorado River the maximum ammonia concentration would increase to just over 2 mg/L approximately 10 years after the flood. Effects of the tailings inundation would then decline over a period of approximately 20 years.

End of current text

8.0 Ground Water Compliance Strategy

The framework defined in the final PEIS for the UMTRA Ground Water Project (DOE 1996) governs selection of the final strategy to achieve compliance with the EPA ground water cleanup and protection standards. This section presents the selection process used to determine the appropriate ground water compliance strategy for the Moab site.

8.1 Compliance Strategy Selection Process

The PEIS framework used to determine the appropriate ground water compliance strategy for the Moab site is summarized in the flow chart provided as Figure 8–1. The framework takes into consideration human health and environmental risk, stakeholder input, and cost. A step-by-step approach is followed until one or a combination of the following three general compliance strategies is selected:

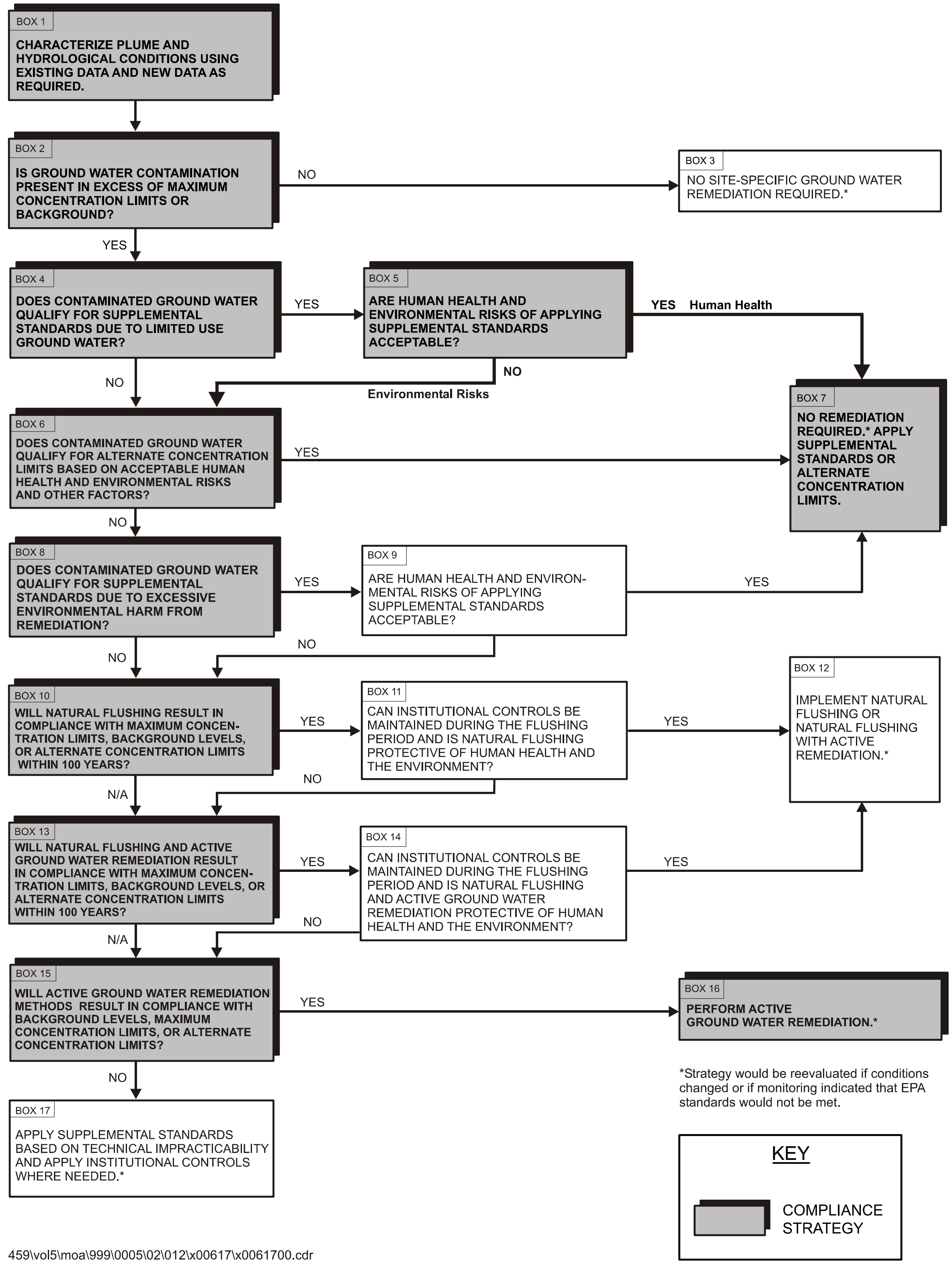
- ***No remediation***—Compliance with the EPA ground water cleanup standards would be met without altering the ground water or cleaning it up in any way. This strategy could be applied for those contaminants that are already at or below maximum concentration limits or background levels, or for those contaminants above maximum concentration limits or background levels that qualify for supplemental standards or ACLs as defined in Section 2.2.1.

- ***Natural flushing***—Compliance with the EPA ground water cleanup standards would be met within a period of 100 years by allowing natural ground water movement and geochemical processes to decrease contaminant concentrations to regulatory limits. The natural flushing strategy could be applied at a site where ground water compliance can be achieved within 100 years, where effective monitoring and institutional controls can be maintained, and where the ground water is not currently and is not projected to be a source for a public water system.

- ***Active ground water remediation***—Compliance with the EPA ground water cleanup standards cannot be met by natural flushing. This option requires application of engineered ground water remediation methods such as gradient manipulation, ground water extraction, treatment, land application, phytoremediation, and in situ ground water treatment to achieve compliance with the standards.

8.2 Ground Water Cleanup Standards

Selection of ground water cleanup actions to meet the EPA standards in 40 CFR 192 is based first on a determination of the appropriate standards for the site and then on identification of a compliance strategy that can meet the standards. Several different ground water standards could apply to an UMTRCA site as described in Section 2.2.1. These include cleanup to background concentrations, maximum concentration limits (Table 2−1), alternate concentration limits, and application of supplemental standards; the applicable standards depend on site-specific cleanup objectives and conditions.



*Strategy would be reevaluated if conditions changed or if monitoring indicated that EPA standards would not be met.

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Figure 8–1. PEIS Compliance Strategy Selection Process

The focus of remediation decisions for UMTRA ground water sites is typically on the uppermost aquifer, which is the aquifer most likely to be affected by uranium-ore processing activities. The uppermost aquifer at the Moab site occurs in unconsolidated Quaternary alluvial material deposited on older bedrock units in the basin that forms Moab Valley. The uppermost aquifer in this portion of the valley is predominantly a highly saline water, often referred to as brine, which can be as thick as 400 ft, topped with a thin layer of less salty water. Estimates are that 60 percent of the aquifer has naturally occurring TDS concentrations greater than 35,000 mg/L. An estimated 82 percent of the aquifer has more than 10,000 mg/L TDS, some of which may be attributed to site-related contamination (see Section 5.5). However, because the ground water in the major portion of the uppermost aquifer has naturally occurring TDS concentrations exceeding 10,000 mg/L, the aquifer meets the definition of a limited use aquifer as described in EPA's *Guidelines for Ground-Water Classification Under the EPA Ground-Water Protection Strategy* (EPA 1988).

Under the provisions of 40 CFR 192, supplemental standards are appropriate for ground water classified as limited use because of naturally poor ambient water quality. Therefore, cleanup of the uppermost aquifer to meet ground water or drinking water standards is not required because limited use aquifers are not likely to be developed as a public drinking water source. Instead, according to EPA regulations in 40 CFR 192. "At a minimum, at sites with limited use ground water, the supplemental standards require such management of contamination due to tailings as is required to assure protection of human health and the environment from that contamination." This means that if site-related contamination could adversely affect a drinking water aquifer or a connected surface water body, management of contamination would be necessary to protect these resources.

For the Moab site, no drinking water aquifer is affected by site-related contamination. Ground water remediation therefore focuses on protecting surface water resources for beneficial uses. Risk calculations show that no unacceptable human health risks would be expected for most probable uses, even using conservative assumptions (see Appendix D of the EIS [DOE 2003b]). However, surface water concentrations exceed aquatic criteria for some site-related constituents. Consequently, the compliance strategy focuses on being protective of the ecological receptors and achieving compliance goals in the surface water.

8.3 Contaminants of Potential Concern

As discussed in Section 5.6, concentrations of several site contaminants in surface water are above appropriate standards or benchmarks for aquatic organisms, including ammonia, copper, manganese, sulfate, and uranium. However, ammonia is the key constituent driving the proposed action due to its high concentrations in the tailings seepage and ground water and its toxicity to aquatic organisms (EPA 1999). As discussed in Section 5.6, federal criteria for the protection of aquatic life have been established for ammonia (EPA 1999), and the State of Utah is in the process of adopting these criteria as state surface water quality standards. Federal criteria have been identified that are protective for both acute and chronic exposures; criteria vary based on temperature and/or pH. Chronic aquatic criteria represent the low end of the potential concentration range for protection of aquatic species from ammonia toxicity—a value of about 0.2 mg/L ammonia (total as N) represents the extreme lower bound of the applicable chronic criteria for the Moab site based on site-specific pH conditions (EPA 1999); the majority of values fall in the 0.6 to 1.2 mg/L range. Acute criteria represent the higher end of the concentration range—the majority of acute criteria for ammonia are between 3 and 6 mg/L.

Available data regarding interaction of ground water and surface water indicate that concentrations of most constituents decrease significantly as ground water discharges to and mixes with surface water (a 10-fold decrease represents a conservative average; see Section 5.6). Therefore, DOE established a target goal for ground water cleanup that would meet acute criteria in surface water with no dilution and would meet chronic criteria assuming a 10-fold dilution. A 3 mg/L target goal for ground water is being used, as this is the low end of the reasonable acute range; with a 10-fold dilution, this would result in compliance with chronic criteria over nearly the entire chronic exposure concentration range. The purpose for establishing the target goal is simply to evaluate ground water modeling results and to assess progress toward compliance. The final remediation objectives are to meet actual acute and chronic standards where they are applicable in surface water.

Ground water modeling results indicate that ground water ammonia concentrations would slowly decline through time under all surface remediation scenarios. With a no-action surface disposal alternative, the lowest ammonia concentrations in the ground water would decrease to 6 mg/L in about 200 years and would therefore not meet the target goal. For the on-site disposal alternative, the target goal would be met after 80 years; for the off-site disposal alternative, the target goal would be met after 75 years.

8.4 Compliance Strategy Selection Process

The PEIS framework shown in Figure 8–1 and site-specific data collected through site characterization and analysis were used to evaluate compliance strategies for Moab site ground water. The compliance strategy selection process for the Moab site based on the current understanding of the site and compliance objectives is summarized in Table 8–1.

Following the PEIS framework, as presented in Figure 8−1, and given the site specific conditions of the Moab site described in previous sections, DOE determined that a compliance strategy employing “no remediation” and the application of supplemental standards to ground water is appropriate for protection of human health; active remediation to prevent unacceptable environmental risks in surface water (i.e., the Colorado River) would be required for all surface alternatives. Under a no-action surface alternative, the active remediation system would need to operate in perpetuity; under the on-site and off-site disposal alternatives, ground water modeling results indicate that the system would have to operate for 80 years and 75 years, respectively.

8.4.1 Proposed Ground Compliance Strategy

The proposed action for ground water is active remediation of ground water contamination to prevent further degradation of surface water quality. This active remediation would be conducted in conjunction with the application of supplemental standards. The focus of the active remediation would be on controlling ground water discharge to potentially sensitive surface water areas, as opposed to accelerating mass removal from the aquifer, though it is anticipated that the remediation should enhance the cleanup process. The proposed active ground water remediation would involve interception of ground water before it enters the surface water, thereby providing plume containment. In addition, ground water injection and/or surface application of clean water from the Colorado River may provide a prompt and continued source of uncontaminated water to the margins of the river where potential contaminant exposure could be the greatest. Monitoring will be ongoing to ensure compliance with surface water standards (see Section 8.4.2 for proposed monitoring approach).

Table 8–1. Summary of Compliance Strategy Selection Process

Box (Figure 8−1)	Action or Question	Result or Decision
1	Characterize plume and hydrological conditions.	The most recent conceptual model of the site is described in Sections 5 and 6 based on characterization activities conducted by DOE in 2002 and 2003. **Move to Box 2.**
2	Is ground water contamination present in excess of 40 CFR 192 MCLs or background?	**YES:** Maximum ground water concentrations of arsenic, cadmium, molybdenum, nitrate, radium, selenium, uranium, and gross alpha exceed the 40 CFR 192 MCLs or other EPA standards at one or more monitoring points. Other constituents such as ammonia and sulfate are elevated compared with background and exceed risk-based concentrations. **Move to Box 4.**
4	Does contaminated ground water qualify for supplemental standards due to classification as limited use ground water?	**YES:** The uppermost aquifer is predominantly composed of brine with concentrations of TDS in excess of 10,000 mg/L, which meets one of the criteria for limited use ground water (60 FR 2861 and EPA 1988). EPA (1988) also indicates that “the entire ground-water unit being classified does not necessarily have to meet Class III [limited use] untreatable criteria, but a major volume would.” The major volume of the uppermost aquifer meets limited use criteria. **Move to Box 5.**
5	Are human health and environmental risks of applying supplemental standards acceptable?	<u>Human Health Risks</u>: **Yes** Ground water is not reasonably considered to be a potential drinking water source because of its limited use designation, and this use of water does not need to be considered further. Initial human health risk assessment results indicate that there are no unacceptable human health risks associated with uses of ground water other than drinking water (e.g., irrigation) and probable uses of hydraulically connected surface water (mainly recreational use). Therefore, protection of human health does not require any cleanup of ground water. For human health, no remediation is required. Apply supplemental standards. **Move to Box 7** (Note: Remainder of compliance strategy selection is focused on environmental risks.) <u>Environmental Risks:</u> **No** Toxicity tests conducted on fish using site-influenced ground water and surface water indicate that there is a potential for adverse effects to aquatic life (Fairchild et al. 2002). Federal criteria for protection of aquatic life have been exceeded for ammonia. Other constituents in surface water are elevated above background levels (e.g., uranium, sulfate). **Move to Box 6.**
6	Does contaminated ground water qualify for alternate concentration limits based on acceptable environmental risks and other factors?	**Not applicable.** Ground water qualifies for supplemental standards. Only surface water concentrations need to be addressed. **Move to Box 8.**
8	Does contaminated ground water qualify for supplemental standards due to excessive environmental harm from remediation?	**No: Move to Box 10.**
10	Would natural flushing result in compliance with 40 CFR 192 MCLs, background, or ACLs within 100 years?	**Not applicable.** Ground water qualifies for supplemental standards. Only surface water concentrations need to be addressed. **Move to Box 13.**
13	Would natural flushing and active ground water remediation result in compliance with 40 CFR 192 MCLs, background, or ACLs within 100 years?	**Not applicable.** Ground water qualifies for supplemental standards. Only surface water concentrations need to be addressed. **Move to Box 15.**
15	Would active ground water remediation result in compliance with background levels, maximum concentration limits, or alternate concentration limits?	**YES:** Active remediation of ground water to control discharge to surface water can achieve surface water remediation goals until natural processes have reduced ground water concentrations to acceptable levels for discharge to surface water. **Move to Box 16.**
16	Perform active ground water remediation.	This is the compliance strategy identified by the PEIS framework.

Specific requirements of the proposed action for active ground water remediation would depend on whether the decision is made to cap the tailings pile in place (on-site disposal) or to relocate the pile to a different site (off-site disposal). The major differences between the two surface options would be the length of time for which an active ground water remediation system would be required to be in operation and the ultimate cleanup goals that can be reached. Treatment alternatives for the active ground water remediation system are described in Section 9.0

8.4.2 Proposed Compliance Monitoring Approach

This section describes the proposed monitoring approach for ground water and surface water at the Moab site. The proposed monitoring serves several purposes. Monitoring will be performed on an as-needed basis (i.e., nonroutine monitoring) to monitor performance of the active remediation system. Part of this monitoring will likely include analyses of samples from paired well and surface water locations to assess the effectiveness of the active remediation system in achieving remediation objectives. This monitoring should also enhance the understanding of ground water and surface water relationships. Other performance monitoring could involve analysis of samples from extraction wells, monitoring of water levels, and other relevant parameters. Monitoring of other wells is also anticipated for the purposes of evaluating attenuation rates of the legacy plume for comparison with model results and assessing progress toward meeting target goals; this type of monitoring would be done on a more routine basis.

Two types of surface water monitoring are expected to take place on a routine basis—general river water quality monitoring and habitat monitoring. It is anticipated that the general river water quality monitoring would be conducted using fixed stations and would involve sampling a flowing portion of the river. This type of sampling would indicate overall water quality of the river. Existing sample locations would be used to the extent possible for background samples and samples adjacent to the tailings pile. A location would also be selected to represent the downstream end of the mixing zone as defined in State of Utah Rule R317-2.

Habitat sampling is proposed to monitor quality of water in areas considered most suitable for young-of-year and larval pikeminnow. These areas may shift location with changing river stage and channel configuration, but monitoring locations should all conform with minimum selection criteria that characterize them as suitable habitat. It is anticipated at this time that the criteria be consistent with those used as the sampling protocol for the Interagency Standardized Monitoring Program for Colorado pikeminnow (Trammel and Chart 1998) and that areas sampled should represent “true backwaters.” At the least, surface water at these locations must cover an area of at least 323 ft^2 and be a minimum of 12 inches deep. Additional temperature and turbidity characteristics may also apply. The number of habitat sample locations identified for each routine round of monitoring will be dependent on time of year, changes in river structure, and water level. Details of sampling methods, locations, and frequency of routine monitoring will be determined during consultation with the appropriate agencies. Nonroutine monitoring requirements for performance assessment of the active remediation system will be determined on an as-needed and ongoing basis as operating parameters are adjusted and optimized.

DOE may also monitor river flows to conduct “opportunistic” sampling of the potentially suitable habitat areas discussed in Section 5.0. When river flows are of sufficient volume for water to occupy areas designated as A, B, and C (Figure 4−23), sampling could be performed to determine site effects on the most suitable habitat areas at the site.

9.0 Screening Level Analysis for Ground Water Remediation Alternatives

This section presents a preliminary screening of technologies for treatment of ammonia- and TDS- contaminated ground water at the Moab site. This is a preliminary screening only and is focused strictly on treatment technology evaluation. No decision has been rendered thus far concerning the need for either surface or ground water restoration at the site. In addition, up to this point no evaluations have been made to determine the feasibility or benefit of ground water restoration. However, if ground water extraction is required at some future time at the site as part of sitewide cleanup, treatment technologies would become necessary. The need for treatment technologies would depend in part on the discharge option selected (assuming ex situ treatment) or whether in situ treatment is more appropriate.

Ground water quality at the Moab site and treatment technologies selected at other UMTRA Ground Water Project sites are discussed in Section 9.1. Effluent discharge options and treatment goals are discussed in Section 9.2. The remainder of Section 9.0 presents a discussion of the possible treatment technologies. These technologies were prescreened for feasibility on the basis of ammonia and TDS as the contaminants of concern and proposed treatment goals for these contaminants (according to the water discharge options).

If the need for ground water treatment is identified during preparation of the Remedial Action Plan for the site, a more comprehensive consideration of ground water chemistry and detailed hydrologic evaluation will be necessary to optimize design and operation parameters for the selected remediation system. That optimization would also require the identification of a final remediation decision for the tailings pile. Data collected to support system design preparation and implementation would be included as part of the Remedial Action Plan.

9.1 Water Quality and Contaminants of Concern

Ammonia is the primary ground water contaminant at the Moab site and occurs above and below the saltwater interface (Section 5.0). Ground water below the saltwater interface also has high naturally occurring TDS concentrations. Typical ground water composition above and below the saltwater interface is summarized in Table 9−1, on the basis of samples from monitor well SMI-PW-01 located between the toe of the tailings pile and the Colorado River. This single well does not characterize the entire aquifer, and it is only considered as representative in terms of evaluation of potential treatment technologies.

Table 9–1. Typical Chemical Composition of the Moab Ground Water Plume (mg/L)

Constituent	Above Saltwater Interface (30−31 ft)	Below Saltwater Interface (55−56 ft)
Ammonia	800	1,600
Sulfate	8,800	14,500
TDS	13,500	46,000
Uranium	2	3

Water quality from the Moab plume is comparable with water quality of four other DOE sites (Shiprock, New Mexico; Tuba City, Arizona; Monument Valley, Arizona; and New Rifle, Colorado) that also have ground water contamination resulting from former uranium milling operations. Treatment technologies selected at these other DOE sites are presented in Table 9−2. These technologies may also be applicable for treatment of the Moab ground water plume.

Table 9–2. Treatment Technologies Selected at Other DOE Sites with Comparable Ground Water Contamination

Shiprock, New Mexico, Terrace	**Tuba City, Arizona**	**New Rifle, Colorado**	**Monument Valley, Arizona**
Evaporation	Distillation	Zero-valent Iron	Phytoremediation

9.2 Preliminary Discharge Options and Treatment Goals

The need for and the degree of treatment required is directly dependent on the discharge option selected for the effluent. Four preliminary discharge options have been identified for the Moab site:

- Discharge to surface water. The Colorado River is a boundary to the Moab site and is the natural discharge point of shallow ground water at the site. It is likely that this option would require extensive water treatment for all contaminants of concern. If discharge to the river is considered a viable alternative for dealing with treatment effluent, appropriate permits would need to be obtained from the state, and compliance with conditions such as discharge rates and effluent composition would be required.
- Evaporation. Solar evaporation would consist of pumping the water into large double-lined outdoor ponds built in the floodplain to withstand 100-year precipitation and flood events. Influent rates to the ponds would match the rate of natural evaporation. Evaporation treats extracted ground water by allowing the water to evaporate due to the dry conditions of the site and warm temperatures during part of the year. Nonvolatile contaminants are contained and allowed to concentrate for later disposal. Evaporation may also be used to treat concentrated wastewater from treatment processes such as distillation and ion-exchange that produce a wastewater stream. Passive evaporation would not require any mixing after disposal in the ponds. However, evaporation rates could be increased considerably by devices such as spray nozzles, at additional costs. This treatment option would require provisions for disposal of the solids accumulated in the evaporation ponds.
- Deep well injection. Most of the site is underlain by the Paradox Formation, which contains thick sequences of salt beds that provide hydrologic isolation from the upper basin-fill aquifer. Due to the relatively high concentration of salts (mostly sulfate and chloride) in the extracted water, there is a possibility that extracted water (with or without pretreatment for one or more constituents) could be disposed of by deep well injection into permeable units that are interbedded within the Paradox Formation. Injection into deeper portions of the saline ground water in the basin-fill aquifer may also provide a zone of isolation. If a geologically isolated zone were found, it is likely that no treatment for any constituents would be needed other than geologic isolation and natural attenuation. This process would

likely require an Underground Injection Control Permit from the state of Utah and concurrence from NRC.

- Shallow injection. Injection wells would be used to return the treated ground water directly back into the alluvial aquifer. Treated ground water could potentially be used to recharge the aquifer at different points to allow manipulation of hydraulic gradients. This could facilitate extraction of the lower quality water and faster removal of the contaminant source. This option would require treatment of ammonia.

Applicable water quality standards for the contaminants of concern will depend on the water discharge option. Potentially applicable water quality standards and guidelines for ammonia and TDS are shown in Table 9–3. Evaporation and deep injection require no contaminant reduction and therefore would not require treatment prior to discharge. Those discharge options are not considered further in this evaluation, though other factors must be considered in evaluating the feasibility of those discharge options. Preliminary treatment goals are provided in Table 9–3 for the remaining discharge options. The purpose of these proposed treatment goals is to provide an order-of-magnitude estimate of the acceptable contaminant levels that the treatment technologies will be required to achieve, according to the different discharge options. It is likely that more specific treatment goals will be available in the future, depending on results of risk analysis or regulatory requirements.

For in situ treatment, it has been assumed that treatment goals will be similar to the option of discharge to surface water, because under an in situ treatment scenario, the Colorado River would be the natural destination of the ground water flow.

Table 9–3. Reported Water Quality Standards, Guidelines, and Proposed Treatment Goals

Discharge Option	Contaminant	Standards	Proposed Treatment Goal
Discharge to the Colorado River	Ammonia	0.2−58 mg/L (aquatic standard based on pH and temperature range; see Appendix D, calculation X0047701)	3 mg/L (target goal; see Section 5.6.5)
	TDS	1,200 mg/L (agricultural use) (SMI 2001)	1,200 mg/L
Evaporation	Ammonia	None	None
	TDS	None	None
Deep injection	Ammonia	None	None
	TDS	None	None
Shallow injection (aquifer flushing)	Ammonia	None	3 mg/L
	TDS	None	None

9.3 Potential Treatment Technologies

Potential technologies evaluated in the screening for the Moab site are listed in Table 9–4. These treatment technologies have been considered at other DOE sites or have been identified as feasible technologies after a literature evaluation.

Table 9–4. Preliminary List of Technologies to Evaluate at the Moab Site

Technology[a]	Description
In Situ Treatment	
Phytoremediation	Plant uptake of nitrogen as a nutrient from the ground water in the root zone. Limited to shallow ground water.
Permeable Reactive Barriers (such as zero-valent iron, air sparging)	Flowing ground water reacts with materials placed in the barrier.
In situ stabilization	Formation of a solid matrix to entrap the contaminants and retard migration in ground water.
Ex Situ Treatment	
Distillation	Sequential evaporation and condensation of the water. The residue is a concentrated brine that can be further concentrated by evaporation.
Coagulation/flocculation with metal hydroxides (iron or aluminum)	Addition of metal salts that precipitate as metal hydroxide and sorb ions in solution.
Ion exchange	Materials that have high affinity for certain solutes. In ion exchange, some ions are exchanged for others. Materials are generally ion-specific.
Chemical oxidation (strong oxidation for ammonia treatment)	Technology used for reduced substances (organic matter, reduced inorganics). Often used as a polishing step for ammonia treatment after a first ammonia treatment stage. Different chemicals might be used (chlorine, ozone, potassium permanganate, hydrogen peroxide).
Chemical reduction (zero-valent iron)	Reducing technology based on metal iron particles. The iron is oxidized to iron hydroxides, and water constituents are reduced.
Biodegradation (nitrification-denitrification)	Proven innovative ammonia treatment technology. It consists of oxidation of ammonia to nitrate/nitrites (using a carbon source); then reduction of these to nitrogen gas. Currently being used at all scales for municipal and industrial water treatment.
Membrane separation (reverse osmosis)	Use of membranes at high pressure to achieve molecular-level separation.
Sulfate coagulation	Technology used for sulfate treatment, limited by the solubility of calcium or barium sulfates.
Ammonia stripping	Proven technology for ammonia treatment.
Ammonia recovery (as magnesium ammonia phosphate)	Innovative technology, currently used at all scales for treatment of industrial and agricultural waste water.

[a] This list does not include combinations of technologies.

9.3.1 Preliminary Evaluation Process for Alternative Treatment Options

The list of technologies shown in Table 9–4 was prescreened. The process involved determining if treatment was necessary for the two identified constituents—ammonia and TDS (i.e., are ground water contaminant concentrations higher than the treatment goals?). The technologies were then evaluated for feasibility (i.e., can this technology treat the ground water and decrease the contaminant concentrations to or close to the treatment goal?). Finally, costs were considered qualitatively (i.e., is the cost of this technology competitive with others that seem feasible?). A diagram of this proposed prescreened process and the resulting qualifiers is shown in Figure 9–1.

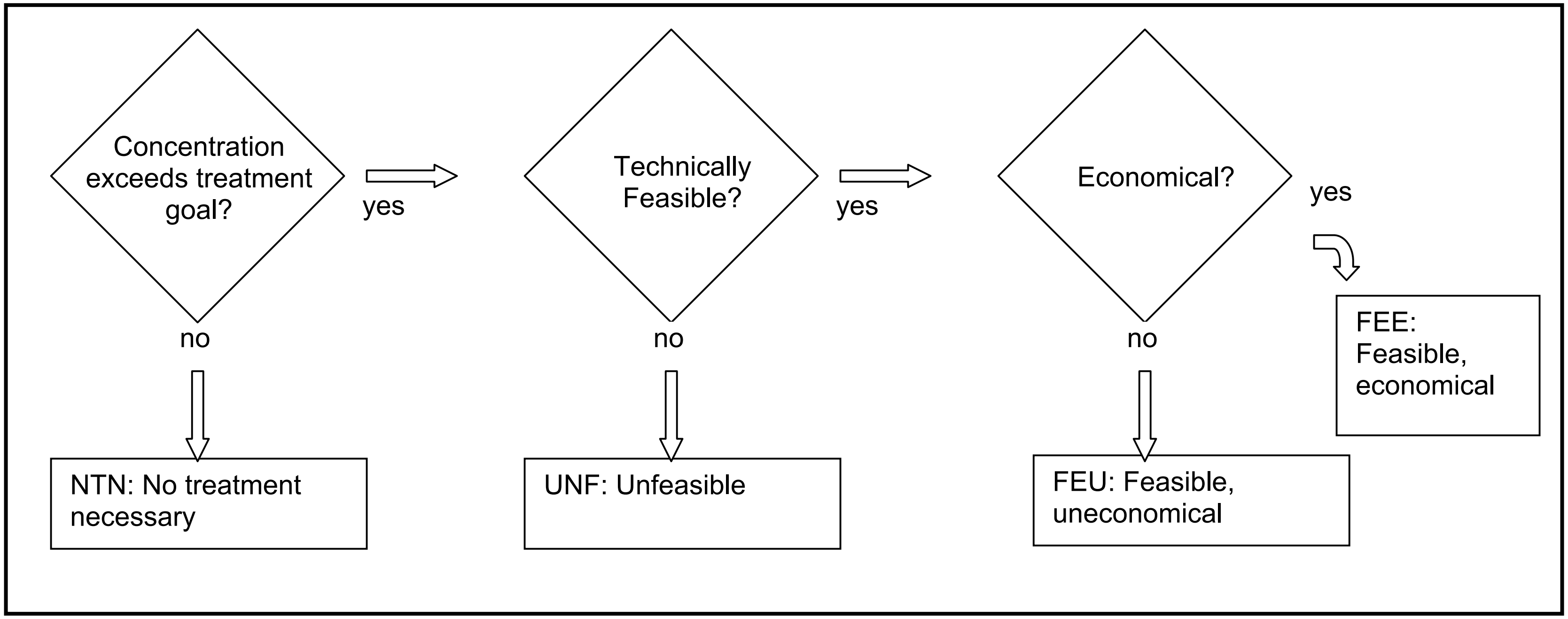

Figure 9–1. Proposed Screening Method for Treatment Options

The resulting qualifiers from this prescreening process are:

- No treatment necessary (NTN): the treatment goal is higher than the ground water concentration.
- Unfeasible (UNF): not capable of meeting treatment goal for the contaminant of interest (i.e., no reports of successful treatment to the preliminary treatment goal have been identified).
- Feasible, uneconomical (FEU): capable of meeting treatment goal for the contaminant of interest, although other alternatives that exist to treat this contaminant are more economical.
- Feasible, economical (FEE): capable of achieving a concentration close to the treatment goal for the contaminant of interest, most likely at a price competitive with that of other technologies.

The results of applying this screening method to the technologies listed in Table 9–4 are shown in Table 9–5 for ex situ treatment options and in Table 9–6 for in situ treatment options. The in situ evaluation is strictly based on whether the specified technology can meet treatment goals and not on the feasibility of being able to install an in situ system. That would also be a major factor in evaluating the overall treatment system. The following sections provide a short description of the technologies identified as feasible and economical for both ammonia and TDS.

9.4 Ex Situ Treatment

9.4.1 Distillation

In distillation, a solution is evaporated by heating it to its boiling point and supplying additional heat for evaporation. Recondensation of the water vapor produces solid-free water. Nonvolatile contaminants (such as inorganic salts) remain in solution. A concentrated liquor or brine is produced, which has to be removed from the evaporation chamber. This brine can be further evaporated in ponds.

9.4.2 Ion Exchange

Porous zeolite (clinoptilolite and sepiolite) can be used for selective sorption of ammonia. The ammonia-contaminated water is passed through the porous medium until the medium is saturated with the contaminant. At this point the ammonia-saturated medium can be regenerated by air stripping or chemical treatment (Corbitt 1993; EPA 1993). Full-scale application of this technology for ammonia treatment has been used successfully.

9.4.3 Chemical Oxidation

Ammonia is the most reduced form of nitrogen and is susceptible to chemical oxidation, producing nitrogen gas. Different chemical oxidizers can be used, such as chlorine, ozone, or potassium permanganate. Each chemical oxidizer has specific requirements (e.g., pH, reaction time, required doses). Also, different by-products are generated, depending on the selection of the chemical oxidizer. For example, chlorination of ammonia produces hydrochloric acid (which, depending of the resulting pH, might need further neutralization). Because the cost of the chemical reagents is considered high, this technology is often used as a polishing step for other ammonia technologies that only achieve partial treatment.

Table 9–5. Ex Situ Treatment Options (Pump and Treat)

Discharge Option	Contaminant	Proposed Treatment Goal (mg/L)	Distillation	Ion Exchange	Oxidation	Zero-Valent Iron	Nitrification-Denitrification	Membrane Separation	Sulfate Coagulation	Ammonia Stripping	Ammonia Recovery
Discharge to Colorado River	Ammonia	3	FEE	FEE	FEE	UNF	FEE	UNF	UNF	FEE	FEE
	TDS	1,200	FEE	UNF	UNF	FEU	UNF	FEE	FEE	UNF	UNF
Shallow Injection	Ammonia	3	FEE	FEE	FEE	UNF	FEE	UNF	UNF	FEE	FEE
	TDS	None	NTN	NTN	NTN	NTN	NTN	NTN	NTN	NTN	NTN

Note: NTN = No treatment necessary.
UNF = Unfeasible.
FEU = Feasible, uneconomical.
FEE = Feasible, economical.

Table 9–6. In Situ Treatment Options

Contamination	Treatment Goal	Phytoremediation	Permeable Reactive Barriers (zero-valent iron)	Permeable Reactive Barriers (sulfate reducing bacteria)	Stabilization
Ammonia	4 mg/L	FEU	UNF	UNF	UNF
TDS	1,200 mg/L	UNF	UNF	UNF	UNF

Note: UNF = Unfeasible
FEU = Feasible, uneconomical

9.4.4 Biodegradation (nitrification-denitrification)

High strength ammonia wastewater has been successfully treated with the combination of anoxic ammonium oxidation (Anammox) and partial nitrification process (SHARON process). The two-stage biological process first oxidizes ammonia to nitrite and nitrate under anaerobic conditions, then reduces nitrate and nitrite to nitrogen gas. These proprietary technologies have been developed recently and applied to large-scale waste (Van Dongen et al. 2001; Hellinga et al. 1998). The process is considered reliable, although it requires careful process control, mostly with respect to the oxygen concentrations and carbon source added for energy. It also has been shown that this technology can be limited by high salt concentrations (Catalan et al. 1997).

9.4.5 Ammonia Stripping

Ammonia is volatile. Mixing ammonia-containing water with ambient air (aeration) will result in ammonia volatilization (stripping). Ammonia stripping has been used for ammonia control under different conditions. Packed columns (stripping towers) or ponds have been used for ammonia stripping, but they present operational problems due to recarbonation (dissolution of ambient carbon dioxide into the water, which causes scale formation). Therefore, packed towers are not typically used in the water treatment industry. Instead, aeration ponds are more widely used, which are simpler to operate and can achieve high ammonia removals. Ammonium is an ion that is in equilibrium with ammonia, the uncharged, volatile form. This equilibrium is controlled by the acid-base conditions of the aqueous solution (characterized by pH). Due to this acid-base equilibrium, ammonia stripping requires pH adjustment to pH 9−10.5.

9.4.6 Ammonia Recovery

The precipitation of ammonia and phosphate in the form of struvite (magnesium ammonium phosphate, or $MgNH_4PO_4 \cdot 6H_2O$ has been considered to treat numerous wastewaters, including drug-processing, yeast production, and tannery waters (Altinbas et al. 2002; Tunay et al. 1997). The process consists of addition of near-stoichiometric amounts of magnesium and phosphate salts (most likely as magnesium hydroxide and sodium phosphate), using a pre-adjusted pH between 8 and 10, at which solubility of the struvite is the lowest. After precipitation, the formed solids are separated by a solid-liquid separation process, such as media filtration (contact clarification) or flocculation-sedimentation. This option would require purchase of process chemicals (magnesium and phosphate salts, as well as chemicals required for pH adjustment), and considerations for waste disposal. Also, this technology does not provide complete ammonia removal. Therefore, it is likely that one or more polishing steps with another ammonia-treating technology would be required.

9.5 Description of Technologies Prescreened as Feasible for TDS Treatment

The following paragraphs briefly describe the technologies from Table 9–5 and Table 9–6 that have been identified as feasible (for at least one of the water discharge options) in the prescreening process for TDS treatment.

9.5.1 Distillation

This option is described in Section 9.4.1.

9.5.2 Membrane Separation (Reverse Osmosis)

Reverse osmosis separates a solute from solution by forcing the solvent to flow through a membrane that is impermeable to the solute, by applying a pressure greater than the osmotic pressure of the solution (which depends on the solute concentration) (Perry and Chilton 1973). Reverse osmosis has frequently been used to desalinize sea and brackish water (with TDS concentrations up to 33,000 mg/L). Reverse osmosis is preferred over other membrane technologies such as electrodialysis, electrodialysis reversal, and nanofiltration for applications where TDS is higher than 3,000 mg/L. Reverse osmosis membranes are permeable to ammonium because of its low molecular weight (ENSR 1997). As a result, ammonium would not be retained in the concentrate and would remain in the treated water. Thus, although a high efficiency of reverse osmosis would be expected for uranium, TDS, and sulfate, reverse osmosis by itself would not be an efficient process for ammonia treatment. Another stage would be required for ammonia removal.

9.5.3 Sulfate Coagulation

This treatment option could reduce TDS by reducing sulfate concentrations only. However, as shown on Table 9–1, sulfates are a significant part of the TDS (65 percent above the saltwater interface and 32 percent below the saltwater interface). Thus, sulfate coagulation has been considered a potentially feasible treatment option for TDS. This technology consists of addition of barium or calcium salts and precipitation of barium or calcium sulfate, respectively. Barium sulfate is very insoluble (about 1,000 times less soluble than calcium sulfate) (EPA 2002). At concentrations of sulfate higher than 1,500 mg/L, addition of calcium (as lime) is preferred, since barium is considered very expensive (Westgeneral 2003). However, lime coagulation has the disadvantage of generating large amounts of sludge.

9.6 Summary

Ground water treatment at the Moab site, if required, could be conducted in situ or ex situ. For ex situ treatment, a preliminary screening resulted in identifying four potential discharge options for ground water: discharge to surface water, evaporation, deep well injection, and shallow injection. Of these options, discharge to surface water and shallow injection are assumed to require some level of ground water treatment. Screening of potential treatment technologies was conducted to present a range of possible technologies that might be considered if ground water remediation is required. Several technologies were identified as both feasible and economical for treatment of the levels of ammonia and TDS associated with the Moab site. The river discharge option appears to be most limiting because of the need for treating both ammonia and TDS; different technologies could be required to address each of these constituents. Development of a formal screening of treatment technologies would be conducted as part of the Remedial Action Plan for the Moab site; this formal screening process would proceed after a preferred alternative is identified for management of the mill tailings at the Moab site.

10.0 Limitations of the Site Conceptual Model

A site conceptual model is a simplification of observed site features that still captures the main complexity, characteristics, and physical aspects of the site. The current site conceptual model honors all the observed site conditions and, it is presumed, contains no inherent bias. The site conceptual model was updated to reflect all the data collected through recent site characterization efforts (Section 5.0); therefore, it represents an updated interpretation of earlier site conceptual models proposed by NRC (1998) and SMI (2001) that has been developed by using the observational approach.

The NRC conceptualization did not account for the contributions of fresh water from surrounding bedrock aquifers. It also did not account for the deep penetration of ammonia into the underlying brine zone below the saltwater interface. The conceptualization proposed by SMI incorporated the contributions of fresh water from bedrock aquifers, but the spatial distribution of the sources differed from what is understood about the site today. For example, the subcrop of bedrock aquifers, which was based on inferred locations of the Moab Fault and the so-called "arcuate fault," were obtained from the maps prepared by Doelling (1985). Recent site characterization efforts (Section 5.0) have shown that the arcuate fault and the buried bedrock terrace probably do not exist. Therefore, the current site conceptual model was updated to reflect the new interpretation. Similarly, recent site characterization has revealed that approximately 60 percent of the ammonia plume lies beneath the saltwater interface.

This section summarizes how the current site conceptual model incorporates the findings from recent site characterization. It then identifies the flow and transport processes that pertain to the management of the tailings pile and ground water remediation alternatives, and lastly makes recommendations that could help further refine the current site conceptual model. Additional revisions to the site conceptual model would be expected in the future as the observational approach continues to reveal new aspects of the site that were previously unrecognized.

10.1 Current Site Conceptual Model

Much of the recently completed work regarding Moab site ground water has focused on improving the understanding of the brine zone and on better defining the distribution of site-related contamination within the aquifer. Important findings include:

- The brine zone is more widespread than previously thought and plays a key role in controlling the distribution of site-related contamination, particularly ammonia.
- There are two ammonia plumes associated with the tailings pile—a deep one below the pile and a shallower one at the southwest toe of the pile. Previously, only the shallow plume had been identified. According to revised estimates, 60 percent of the ammonia mass resides below the saltwater interface and 40 percent lies above it.
- Highest concentrations of ammonia contamination in the ground water beneath the pile are actually in the brine zone. The vertical extent of the ammonia plume beneath the pile has not been delineated. Ammonia moves upward from the brine zone through advective, dispersive, and diffusive processes, which occur relatively slowly.
- The deep ammonia plume was probably the result of density-driven flow as high TDS tailings fluids seeped from the tailings pond.

- The interface between fresh water and salt water was determined more accurately, and therefore, the thicknesses of the freshwater and saltwater zones are now known more accurately. Approximately 82 percent of the ground water in the basin-fill aquifer contains TDS concentrations greater than 10,000 mg/L, and about 60 percent of the aquifer contains salt water (TDS concentrations greater than 35,000mg/L).
- Pumping can be used to extract ground water above the saltwater interface; through judicious positioning of the pumps along the well screen and carefully controlled pumping rates, the vertical movement of the interface can apparently be controlled.
- The vertical and horizontal position of the brine interface is sensitive to changes in elevation of the Colorado River.
- The investigation of the tailings pile and chemistry of pore fluids in the pile identified an evaporative salt layer associated with 18,000 mg/L ammonia concentrations in the upper portion of the tailings pile. This layer represents a potential source of future contamination through dissolution and seepage to the ground water below.
- Ammonia loading to the river is from three distinct sources in the alluvial aquifer: downward seepage of tailings pore fluids to the alluvial aquifer, lateral migration of the ammonia plume in the alluvial aquifer above the saltwater interface, and upward flux into the fresher water system from the brine zone.

10.2 Limitations of the Current Conceptual Model

10.2.1 DOE’s Interpretation

Despite the improved understanding of the site conceptual model, its increased complexity raises some additional questions. For example, because of the pervasiveness of the brine facies, it now appears that ground water remediation will be driven mainly by ecological concerns regarding surface water quality as opposed to meeting ground water standards. However, understanding of the interrelationships of the ground water and surface water systems, particularly with respect to water quality influences, is still limited. A target level for ground water remediation has been established that is believed to be protective of surface water; however, this goal is conservative, and it is plausible that higher ground water concentrations, which are more readily achievable, could be protective as well.

The ability to meet target goals was evaluated on the basis of results of ground water modeling. Current modeling indicates that remediation goals would be met for either the on-site or off-site tailings pile management options. However, modeling results are most sensitive to a few transport parameters, each of which contain uncertainty due to limited site specific data—decay rate of ammonia, retardation of ammonia transport (represented by K_d), and estimated future contributions of ammonia from the tailings. In addition, the upward migration of ammonia from the brine is not a well-understood process. Although parameter estimates used in the model appear reasonable and are consistent with site observations, errors in any or all of these estimates could mean the remediation objectives might not be achieved. This is particularly true for an on-site disposal alternative because of the continued ammonia seepage from the pile. The tailings seepage source term is eliminated for an off-site disposal alternative, reducing the uncertainty.

Although it is now known that ammonia contamination extends to a significant depth into the brine zone beneath the pile, the vertical extent and mass of site-related contamination beneath the pile has not been fully delineated. Similarly, the time for the ammonia mass in the brine zone to become depleted is not clear. Background concentrations of constituents in the brine zone, particularly trace constituents, are also not well characterized.

10.2.2 State of Utah Hydrogeologic Investigation of the Matheson Wetlands Preserve

There is also some uncertainty about whether site-related contamination extends beneath the river into the Matheson Wetlands Preserve. The location of the saltwater interface and role of the brine and the river as a boundary condition in controlling ground water flow on the Matheson Wetlands Preserve side of the river is not well known. The state of Utah Division of Environmental Quality and the University of Utah have recently completed a hydrogeologic investigation on the Matheson Wetlands Preserve to provide greater understanding of these aspects; however, the results were not available to incorporate into the current site conceptual model in this version of the SOWP. Should results from the Utah investigation warrant changes to the site conceptual model, they will be made in future versions.

10.3 Summary and Conclusions

In their review of DOE’s Plan for Remediation for the Moab site, the National Academies of Science (NAS 2002) indicated that the technical basis at that time was not adequate to support a tailings disposal decision and that additional data and analysis were needed. Data collection and analysis performed in preparation of this SOWP have focused primarily on improving the technical knowledge of the site and focused on areas specifically identified by the NAS related to surface water and ground water, as well as supporting selection of a ground water compliance strategy for the site. Results of site characterization have indicated that ground water at the Moab site is not a potential source of drinking water due to naturally high levels of TDS. However, ground water does discharge to surface water and site-related contamination must be addressed in order to be protective of surface water quality. Based largely on the results of ground water modeling performed for the Moab site, it appears that ground water concentrations can be achieved that are protective of surface water quality in the Colorado River with either an on-site or off-site disposal alternative. Regardless of the surface alternative selected, some type of active ground water remediation will be required until protective levels are attained.

As discussed previously, there are uncertainties associated with modeling results. Though the model was developed using site specific data to the extent possible, at the extremes the remaining uncertainties do not eliminate the possibility that target goals may not be achievable or that timeframes are under- or overestimated. However, the consequences associated with incorrect modeling results are probably not significant, particularly once an active remediation system is operational. The target goal established for ground water cleanup (3 mg/L ammonia) is a conservative one. It does not account for dilution effects as ground water discharges to the river and is at the low end of the acute federal criteria range of aquatic criteria. Observed concentrations of ammonia in Colorado River water under current conditions have only exceeded acute aquatic ammonia criteria in limited areas and have not affected overall river water quality. The anticipated three orders of magnitude (1,000 fold) decrease in ground water ammonia concentrations over the next several decades, of which there is high confidence, strongly suggests that the extent of potential future impacts to surface water quality and actual risk to all aquatic species will be extremely limited to non existent even if model predictions are not fully realized. Currently, the major concern for ammonia discharge is for threatened and endangered fish that inhabit the river in the area. Potentially suitable habitat is not well defined in the site vicinity; influence of site-related ammonia on potential suitable habitat is not well known. However, with active ground water remediation, risks to the species should be reduced for both the short-term and long-term. In addition, it is anticipated that the endangered fish will be delisted before active ground water remediation is estimated to end, based on currently increasing fish populations; therefore future discharges of ammonia may be of lesser concern.

While DOE has improved the technical basis on which to make a decision regarding the Moab site, improvements can be always be made, as is the case with any remediation project. More will be learned about the site during operation of the interim action through the observational method which has been proven successful at numerous other Title I uranium mill tailings sites. These data can be incorporated into the existing understanding of the site. For the final active remediation system, more data may be needed to optimize system design and implementation.

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