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Plate 1. Geologic Map and Ground Water Monitor Well Locations a the New Rifle Site

Acronyms and Abbreviations

ACL	alternate concentration limit
AEC	Atomic Energy Commission
ASTM	American Society of Testing Materials
AWQC	aquatic water quality criteria
BLRA	baseline risk assessment
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
cfs	cubic feet per second
cm/yr	centimeters per year
CoC	chain of custody
COC	contaminants of concern
COPC	contaminants of potential concern
CT	central tendency
DOE	U.S. Department of Energy
EHPA	ethylhexylphosphoric acid
EPA	U.S. Environmental Protection Agency
EPC	exposure point concentation
ERA	ecological risk assessment
ET	evapotranspiration
ft	foot (feet)
ft/day	foot (feet) per day
ft^3/day	cubic feet per day
GCAP	Ground Water Compliance Action Plan
GJO	Grand Junction Office
gpm	gallons per minute
HEW	U.S. Department of Health, Education, and Welfare
HI	hazard index
HQ	hazard quotient
i.d.	inside diameter
K_d	distribution coefficient
KW-hr	kilowatts per hour
LOAEL	lowest observed adverse effect level
MAP	management action process
MCL	maximum contaminant level
MDRD	minimum detectable relative difference
µm	micrometer
mg/Kg	milligrams per kilogram
mg/L	milligrams per liter
mL	milliliters
mL/g	milliliters per gram
µg/L	micrograms per liter
mm	millimeters
NEPA	National Environmental Policy Act
NOAEL	no observed adverse effect level
NRC	U.S. Nuclear Regulatory Commission
O&M	operating and maintenance

OMB	Office of Management and Budget
o.d.	outside diameter
pCi/L	picocuries per liter
pCi/g	picocuries per gram
PEIS	Programmatic Environmental Impact Statement
ppm	parts per million
PVC	polyvinyl chloride
RAP	remedial action plan
QRA	quantitative risk assessment
RBC	risk-based concentration
RME	reasonable maximum exposure
RO	reverse osmosis
rpm	revolutions per minute
RR	relative risks
RRM	residual radioactive material
SDWA	Safe Drinking Water Act
SOWP	site observational work plan
SQC	sediment quality criteria
TAGR	Technical Approach to Ground Water Restoration
TDS	total dissolved solids
TRV	toxicity reference value
UCL_{95}	95 percent upper confidence limit
UMTRA	Uranium Mill Tailings Remedial Action (Project)
UMTRCA	Uranium Mill Tailings Radiation Control Act
USGS	U. S. Geological Survey
ZVI	zero-valent iron

Executive Summary

Ground water beneath the New Rifle, Colorado, site was contaminated by former vanadium and uranium ore-processing operations that were ongoing from 1958 through 1973. From 1973 to 1984, part of the mill was used to produce vanadium; this operation did not produce tailings and involved processing vanadium-bearing solutions. Mill tailings and other associated radioactive contaminated surface materials were removed from the site by 1996 in accordance with criteria established in 40 CFR Part 192 Subpart A as part of the Uranium Mill Tailings Remedial Action (UMTRA) Surface Project. However, the potential for infiltration of ground water contaminants remained until that time. It is also possible that residual contamination, particularly from nonradiological constituents, is a potential continuing source of ground water contamination.

Site-specific field investigations reveal the surficial aquifer is the only aquifer affected by the former milling operations. Contaminants of concern (COCs) in the surficial aquifer are identified as ammonia, arsenic, fluoride, manganese, molybdenum, nitrate, selenium, uranium, and vanadium based on potential human health risks. Uranium is the most prevalent site-related contaminant occurring in the surficial ground water. Concentrations exceeding the UMTRA standard occur several miles beyond the downgradient edge of the site. Molybdenum, ammonia, and nitrate plumes also extend beyond the site boundary to just beyond the western edge of the UMETCO property, which adjoins the site on the west. However, these plumes decrease to background beyond this point. Selenium, manganese, vanadium, and fluoride plumes are restricted to only the site and UMETCO properties. Arsenic contamination is confined to the New Rifle site. Maximum concentrations of uranium, molybdenum, selenium, nitrate, and arsenic exceed their respective UMTRA standards. Fluoride exceeds the Safe Drinking Water Act (SDWA) maximum contaminant level (MCL). Manganese, ammonia, and vanadium exceed risk-based concentrations for human health in a residential setting.

The ecological risk assessment indicates that the greatest potential environmental problem will be the revegetation of the mitigation wetland at the west end of the site. A Wetlands Management Plan will be developed to address these concerns.

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

The proposed compliance strategy to cleanup the surficial ground water at the New Rifle Site is no remediation for constituents that do not pose a potential risk and do not exceed EPA standards. For constituents that pose a potential risk or exceed EPA standards or both, the strategy is to perform natural flushing ground water remediation in combination with the establishment of alternate concentration limits (ACLs) and institutional controls. Pending results of a pilot study, the strategy for vanadium may require some type of active remediation.

Numerical modeling of ground water flow and transport indicates that concentrations of arsenic, molybdenum, nitrate, and uranium will decrease to UMTRA standards or background concentrations during the 100-year natural flushing period. The ACL proposed for selenium is the SWDA MCL of 0.05 mg/L. The proposed ACL will be met within the 100-year natural flushing period as demonstrated by ground water flow and transport modeling. While not modeled, it is anticipated that ammonia, fluoride, and manganese will decrease to acceptable levels based on historical trends and modeling of other constituents of similar geochemical characteristics. Fluoride should reach the SDWA MCL of 4.0 mg/L. The proposed ACL for ammonia is 200 mg/L (as NH_4) and is the risk-based level for human health that is protective for residential use of drinking water. An ACL for manganese is based on background and is set at 4.0 mg/L. A risk-based ACL of 0.33 mg/L for vanadium is proposed. Ground water modeling indicates this level cannot be reached in the 100-year natural flushing period. However, observed decreases of vanadium are inconsistent with model results and a pilot study is recommended before making a final remedial decision for this constituent. Institutional controls in the form of deed restrictions and zone district changes will prohibit use of untreated ground water during the period of natural flushing and ground water monitoring will be conducted to observe the progress of remediation. Information presented in this final site observational work plan supports the proposed compliance strategy in a manner that is consistent with the regulatory compliance framework.

1.0 Introduction

1.1 Purpose and Scope

The New Rifle Uranium Mill Tailings Remedial Action (UMTRA) Project site is a former ore-processing facility located approximately 2.3 miles west of the city of Rifle, between U. S. Highway 6 and Interstate 70, in Garfield County, Colorado (Figure 1–1). The site is situated on a broad alluvial floodplain. The site is bounded on the east and south by the Colorado River and on the north side by steeply ascending slopes of the more resistant sedimentary rocks of the Wasatch Formation.

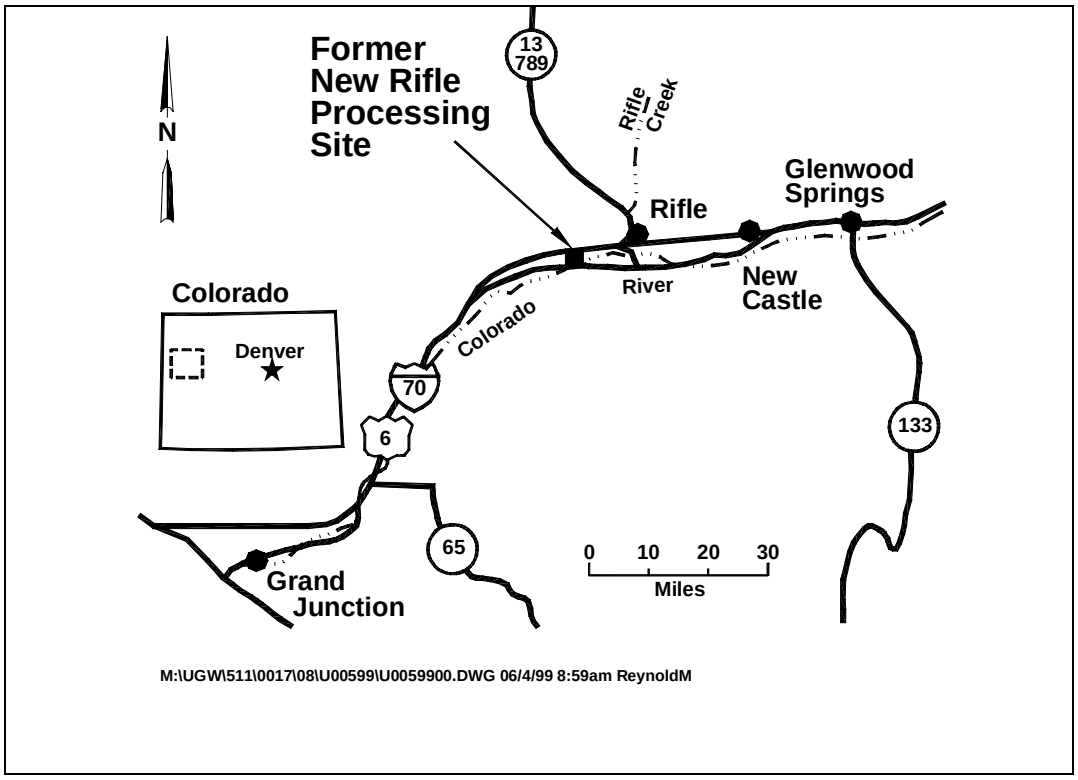

Figure 1–1. Location of the New Rifle Site

The U.S. Department of Energy (DOE) completed surface remediation of abandoned uranium mill tailings and other contaminated surface radioactive material associated with the former milling operation at the site by relocating the contaminated materials to the Estes Gulch disposal cell approximately 9 miles north of Rifle. Surface remedial action began in 1992 and was completed in 1996. The former processing site is currently covered and regraded with clean fill material and reseeded with range grasses.

DOE's goal is to implement a cost-effective compliance strategy that is protective of human health and the environment by remediating contaminated ground water at the New Rifle site. The proposed compliance strategies to cleanup the alluvial ground water at the New Rifle site are (1) no ground water remediation of constituents that do not pose a potential risk and do not exceed U.S. Environmental Protection Agency (EPA) standards, and (2) for constituents that pose a potential risk or exceed EPA standards or both, the strategy is to perform natural flushing

ground water remediation in combination with the establishment of alternate concentration limits (ACLs) for selected constituents. A pilot study for vanadium contamination will be performed to determine if vanadium will naturally flush, or if active remediation will be required. Institutional controls will prohibit some ground water uses during the natural flushing period and ground water monitoring will be conducted to observe progress of remediation.

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

Compliance requirements for meeting the regulatory standards at the New Rifle site are presented in Section 2.0. Site background information, including an overview and history of the former milling operation and current water and land use, are reviewed in Section 3.0. Results of the 1998 field investigations conducted at the site are presented in Section 4.0. Site-specific characterization of the geology, hydrology, geochemistry, and ecology are synthesized in the site conceptual model in Section 5.0. Potential human health and ecological risks associated with ground water contamination are summarized in Section 6.0, and the proposed compliance strategy to clean up the ground water is presented in Section 7.0. The evaluation of alternatives is in Section 8.0.

1.2 UMTRA Project Programmatic Documents

Programmatic documents that guide the SOWP include the *UMTRA Groundwater Management Action Process* (MAP) (DOE 1999b), the *Final Programmatic Environmental Impact Statement for the Uranium Mill Tailings Remedial Action Ground Water Project* (PEIS) (DOE 1996b), and the *Technical Approach to Groundwater Restoration* (TAGR) (DOE 1993b). The MAP states the mission and objectives of the UMTRA Ground Water Project and provides a technical and management approach for conducting the Project. The PEIS is the programmatic decision-making framework for conducting the UMTRA Ground Water Project. DOE will follow PEIS guidelines to assess the potential programmatic impacts of the Ground Water Project, to determine site-specific ground water compliance strategies, and to prepare site-specific environmental impact analyses more efficiently. Technical guidelines for conducting the ground water program are presented in the TAGR.

1.3 Relationship to Site-Specific Documents

The surface remedial action plan (RAP) (DOE 1992) provides early site characterization information. This information was updated in developing the SOWP to strengthen the site conceptual model. After a ground water compliance strategy is selected for this site, a draft and final Ground Water Compliance Action Plan (GCAP) will be prepared to document the remediation decision.

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

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

1.4 SOWP Revisions

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

The draft SOWP was prepared in 1996 and included all previous information about the site, presented a proposed compliance strategy, and defined additional data needs that were required to support the most likely compliance strategy. Following stakeholder review and resolution of comments, fieldwork was conducted in 1998 to address the data gaps identified in the draft SOWP.

This final SOWP presents the additional data collected in 1998, correlates the data to previous information, updates the site conceptual model and risk assessment, and recommends a final compliance strategy based on the most current information.

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 former New Rifle millsite to achieve compliance with Subpart B of EPA health and environmental protection standards for uranium and thorium mill tailings (40 CFR Part 192) and the final rule to the standards published in 60 FR 2854.

2.1 Uranium Mill Tailings Radiation Control Act

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

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

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

2.1.1 EPA Ground Water Protection Standards

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

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

2.1.1.1 Subpart B: Standards for Cleanup of Land and Buildings

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

- Background levels, which are concentrations of constituents in nearby ground water not contaminated by ore-processing activities.

- UMTRA maximum concentration limits, which are limits set by EPA for certain hazardous constituents in ground water and are specific to the UMTRA Project (Table 2–1). [Note that maximum contaminant levels (MCLs) have also been established for some of these constituents by EPA under the Safe Drinking Water Act (SDWA) (also provided in Table 2–1). Some of the UMTRA standards are the same as the MCLs for corresponding contaminants, but differences do exist.]

- ACLs, which are concentration limits for hazardous constituents that do not pose a substantial hazard (present or potential) to human health or the environment as long as the limit is not exceeded.

Table 2–1. Standards for Inorganic Constituents in Ground Water at UMTRA Project Sites[a]

Constituent	UMTRA[b]	SDWA[c]
Arsenic	0.05	0.05
Barium	1.0	2.0
Cadmium	0.01	0.005
Chromium	0.05	0.1
Lead	0.05	N/A
Mercury	0.002	0.002
Molybdenum	0.1	N/A
Nitrate (as N)	10.0[d]	10.0[d]
Selenium	0.01	0.05
Silver	0.05	N/A
Combined radium-226 and radium-228	5 pCi/L	5 pCi/L
Combined uranium-234 and uranium-238	30 pCi/L[e]	N/A
Gross alpha-particle activity (excluding radon and uranium)	15 pCi/L	15 pCi/L

[a]Concentrations reported in milligrams per liter (mg/L) unless otherwise noted.
[b]Maximum Concentration of Constituents for Groundwater Protection, UMTRA Standard (40 CFR Part 192, Table 1, Subpart A).
[C]Maximum Contaminant Levels, Safe Drinking Water Standard (40 CFR 141.23 and 141.62).
[d]Equivalent to 44 mg/L nitrate as NO_3.
[e]Equivalent to 0.044 mg/L, assuming secular equilibrium of uranium-234 and uranium-238.

N/A = not applicable
pCi/L = picocuries per liter.

Natural Flushing to Achieve Standards

Subpart B also allows natural flushing to meet EPA standards. Natural flushing allows natural ground water processes to reduce the contamination in ground water to acceptable standards (background levels, UMTRA MCLs, or ACLs). Natural flushing must allow the standards to be

met within 100 years. In addition, institutional controls and an adequate monitoring program must be established and maintained to protect human health and the environment during the period of natural flushing. Institutional controls would prohibit inappropriate uses of the contaminated ground water. The ground water also must not be a current or projected source for a public water system subject to provisions of the SDWA during the period of natural flushing.

2.1.1.2 Subpart C: Implementation

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

Supplemental Standards

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

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

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

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

- There is no known remedial action.

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

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

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

2.1.2 Cooperative Agreements

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

2.2 National Environmental Policy Act

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

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

3.0 Site Background

The New Rifle UMTRA Project site is located in western Colorado, approximately 2.3 miles west of the city of Rifle (Figure 1–1). The 142-acre site, which is accessible by U.S. Highway 6, is the location of a former vanadium and uranium mill that operated from 1958 through 1984. An overview of the site's physical setting and climate, a history of the former milling operation, a summary of previous investigations, and the city of Rifle's current land and water use plan are presented in the following sections.

3.1 Physical Setting and Climate

The former millsite is located near the northeastern edge of the Colorado Plateau Physiographic Province. Mature, stream-eroded upland pediment surfaces, large structural upwarps and intervening basins, and flat-topped basalt mesas characterize the area. Major features include the Roan Cliffs to the northwest, the Grand Hogback monocline to the north and northeast, and the Colorado River and Taughenbaugh Mesa along the northern flank of the Battlement Mesa to the south. Topographic elevations range from approximately 5,300 feet (ft) above mean sea level at the site to 8,000 ft along the Grand Hogback to the north and to more than 10,000 ft on top of the Battlement Mesa to the south. The region has an arid to semiarid climate with high evaporation, low precipitation, low humidity, and large temperature variations.

The weather station closest to the Rifle site is operated by the National Weather Service (station number 57031) which is located a few miles to the southeast of the city. Climatological data collected from the weather station for the period 1910 through 1997 is presented in Table 3–1 and indicates the site receives on average approximately 11.0 inches of total precipitation per year. Rainfall occurs during the summer in high-intensity, short-duration, late afternoon thunderstorms that are conducive to runoff. Precipitation occurs in the winter as snowfall.

Table 3–1. Meteorological Data for Rifle, Colorado

Month	Air Temperature (°F)			Precipitation (inches)	Potential Evaporation (inches)
	Mean Low	Mean	Mean High		
January	8.9	22.6	36.3	0.83	0.0
February	16.2	29.9	43.6	0.71	0.0
March	24.0	38.6	53.5	0.88	0.62
April	31.3	47.6	64	0.94	1.67
May	38.7	56.2	73.7	0.95	2.98
June	45.1	64.6	84.0	0.70	4.13
July	51.8	70.8	89.9	1.00	5.33
August	50.2	68.9	87.5	1.08	4.6
September	41.3	60.4	79.4	1.05	3.12
October	31.0	49.1	67.2	1.13	1.73
November	21.3	36.3	51.3	0.83	0.25
December	12.4	25.8	39.2	0.91	0.0
Annual	31.0	47.6	64.1	11.0	

Temperatures show considerable diurnal and seasonal variations. Winters are cold, with average low temperatures typically below freezing from December through February. Summers are hot, with average high temperatures up to 90 °F in July.

Potential evaporation (inches) for the Rifle area was estimated based on the Thornthwaite equation (Thornthwaite and Mather 1957) using the site air temperature data presented in Table 3–1 (Calculation Set U0044700). The data indicates potential evaporation exceeds precipitation from April through October. The highest potential evaporation occurs in July when evaporation exceeds 5 inches.

3.2 Site History

Historically, vanadium and uranium ores were processed at two different mill sites located near the city of Rifle, Colorado. U.S. Vanadium Company constructed the first mill in 1924 for the production of vanadium (Merritt 1971). This plant was located approximately 0.3 miles east of the city and is referred to as the Old Rifle site. Union Carbide and Carbon Corporation (Union Carbide) purchased the assets of the U.S. Vanadium Company in 1926 and established the U.S. Vanadium Corporation as a subsidiary (Chenoweth 1982). The subsidiary operated the former Old Rifle plant intermittently until 1946 when it was modified to include the recovery of uranium as well as vanadium. Production continued until 1958 when the old plant was replaced with a new mill located approximately 2.3 miles west of the Old Rifle site. The former location of the new mill is referred to as the New Rifle site (Figure 1–1).

Uranium and vanadium production at the New Rifle mill lasted from 1958 to 1984. Concentrated ore was shipped to the New Rifle mill from 1958 to the early 1960s by truck and railroad from upgrading plants at Green River, Utah, and Slick Rock, Colorado (Merritt 1971). Ore for the Green River concentrator came primarily from southeast Utah; ore for the Slick Rock concentrator came from numerous mines in the Uravan Mineral Belt (DOE 1982). From 1964 to 1967, the New Rifle mill also processed lignite ash produced by Union Carbide's strip mining operations near Belfield, North Dakota. From 1973 to 1984, part of the mill was used to produce vanadium; this operation, which did not produce tailings, involved processing vanadium-bearing solutions from Union Carbide's plant at Uravan, Colorado, for various vanadium products used by the steel industry.

Uranium ore with relatively low-grade vanadium was separated in a direct acid-leaching step. Higher-grade vanadium ores were initially salt roasted. The Atomic Energy Commission (AEC) records document that 2,259,000 cubic yards of Old Rifle tailings and 1,802,019 tons of ore were processed. The AEC purchased 5,852 tons of uranium oxide (U_3O_8) and 2,162 tons of vanadium oxide (V_2O_5) produced by the New Rifle mill (DOE 1982).

The west central portion of the New Rifle mill site contained 33 acres of tailings in two distinct piles. The combined piles measured approximately 1,600 ft in the north-south direction and approximately 1,150 ft in the east-west direction. The northwest pile contained older tailings while the southwestern pile contained the more recent tailings. The tops of both piles were relatively flat; however, the sides were steep with nearly 45° slopes in many places. Process mill buildings were located north and east of the piles. Former holding ponds that held processing wastes (including vanadium and gypsum) were located east of the piles. The location of the tailing piles, evaporation ponds, ore storage area, and mill buildings as they existed in 1974 are shown in Figure 3–1.

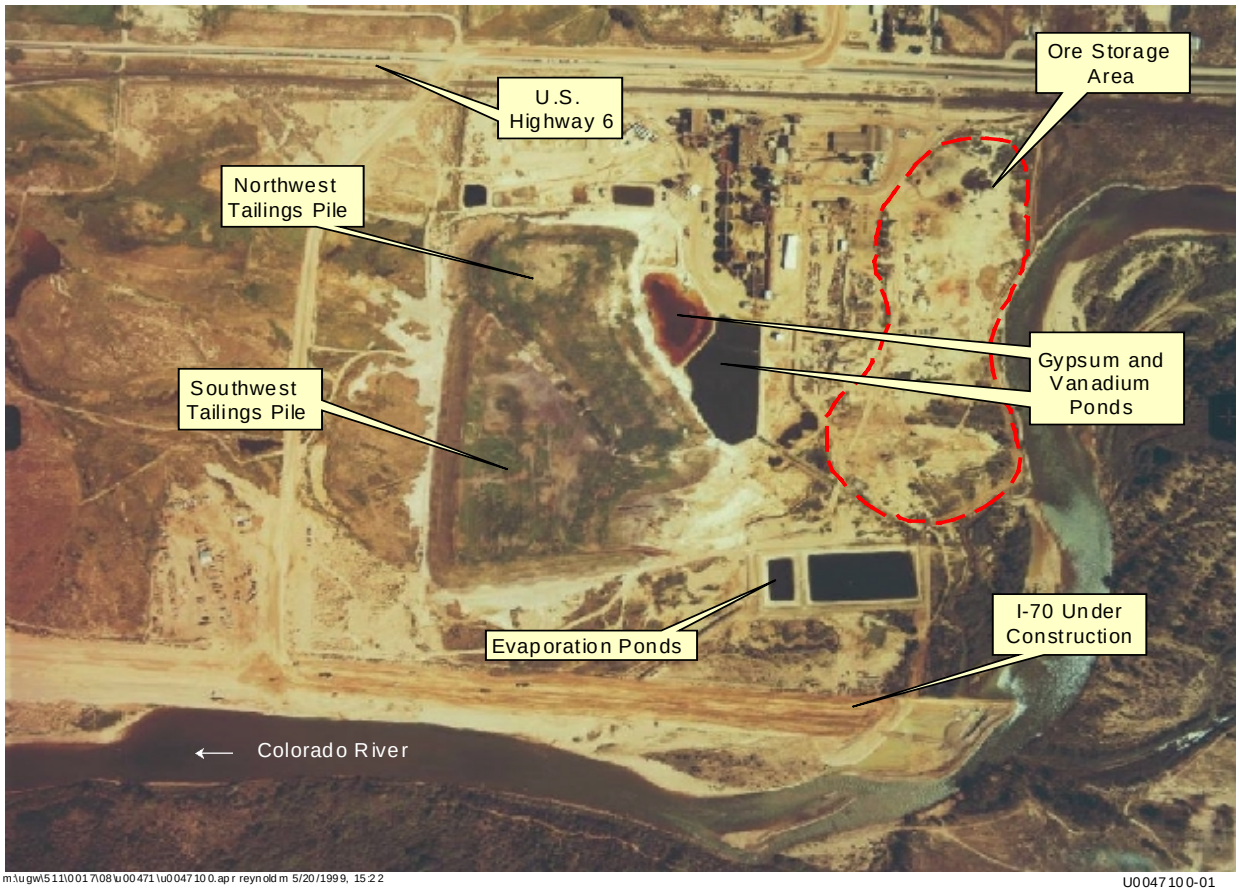

Figure 3–1. New Rifle Mill Site Showing the Location of the Northwest and Southwest Tailings Piles, Holding Ponds, Mill Buildings, and the Ore Storage Area—August 1974

Figure 3–2. View of the New Rifle Site Looking West During the Early Stages of Surface Remedial Action—August 1989

The tailing piles were partially stabilized by Union Carbide with the application of mulch and fertilizer. An irrigation system was installed to promote growth of native grasses that were planted. However, much of the pile did not revegetate, and wind and water eroded some of the tailings. The tailings pile at the beginning phase of the surface remediation in 1989 is shown in Figure 3–2. All tailings, radiological contaminated materials, and associated process buildings and structures were removed from the site during the surface remedial action completed in 1996.

3.2.1 Previous Investigations

The U.S. Department of Health, Education, and Welfare (HEW) (1962) and Merritt (1971) provide detailed descriptions of the uranium concentration process, mill by-products, and process waste streams. Albrethsen and McGinley (1982) summarize the history of the domestic uranium procurement policies and practices under the AEC.

Fischer (1960) and Chenoweth (1982) document the vanadium and uranium deposits located along East Rifle Creek approximately 15 miles northeast of the site. Geologic studies conducted in the area of the site are reported by the U.S. Geological Survey (USGS) (Donnell 1969; Shroba and others 1995) and the Colorado Geological Survey (Stover 1993).

Site-specific hydrogeologic and geochemical investigations are described in an Environmental Impact Statement (DOE 1990), a RAP (DOE 1992), a BLRA (DOE 1996a), and the draft SOWP (DOE 1996c). Organic compounds used at the New Rifle site are addressed in *Phase II Organic Investigation of Ground Water Contamination at the New Rifle Site* (DOE 1997a).

3.3 Land and Water Uses

The population of the city of Rifle is approximately 6,000. The town contains businesses, industrial areas, and residential neighborhoods. A land use map for the New Rifle Site and surrounding areas is presented in Figure 3–3. The site is located just outside the Rifle city limits and is under the jurisdiction of Garfield County. Garfield County currently zones the site, which is located within the 100-year floodplain, for agricultural/industrial use.

Several light industrial facilities or commercial operations, accessible from U.S. Highway 6, are present within 0.25 miles north of the site and include a machine shop, an insulation supply business, and a salvage yard. A bulk oil and gasoline supplier is located at 69 County Road 24 to the northeast.

Several sand and gravel mining operations occur along the floodplain west and east of the site. Roaring Fork Resources operates a gravel mine on property leased from UMETCO approximately 0.50 miles west of the site. The city of Rifle operates a series of wastewater sewage lagoons just east of the site.

There are no residences adjacent to the site. The nearest residence is located approximately 0.25 miles to the north of the site fence line across U.S. Highway 6.

The city of Rifle's future comprehensive growth plan identifies the land within the New Rifle site boundaries zoned "Commercial/Service Cluster." This designation would allow for a western park, county fairgrounds, year-round recreation, education, agricultural facilities, a park trail to the visitor center and downtown, and/or wetland/recreation/wildlife enhancement use at a future

date if the city of Rifle annexes the property. The site is currently deeded to the state of Colorado. Plans to transfer the land to the community are in progress.

Ranching and farming are the main land uses on Prefontaine Mesa north of the New Rifle site. Prefontaine Mesa is also heavily irrigated.

3.3.1 Potable Water Use

Residents in the city of Rifle obtain potable water from the municipal water system, which receives surface water from the Colorado River and Beaver Creek. Water from the Colorado River is collected at an inlet approximately 3.5 miles upstream of the site. Water in Beaver Creek originates from Beaver Mountain snowmelt and precipitation across the river approximately 5 miles southwest of the city and is transported into the municipal water supply via a pipeline. Beaver Creek supplies approximately 10 percent of the city's municipal water.

The city provides potable water to some users outside the city limits, though most residents living outside the municipal boundaries obtain water from private wells or springs (DOE 1996a). Natural ground water quality in the Colorado River alluvium and weathered bedrock is poor tasting and considered unpalatable because of high sulfate and TDS levels. Consequently, the ground water from these private wells and springs generally is not used for drinking water unless treated. Ground water is used for other domestic purposes such as bathing, watering domestic livestock, and watering gardens. The nearest residential well (location 442) is located approximately 1.5 miles downgradient (west) of the site. This well uses a reverse osmosis system to treat the ground water to an acceptable quality.

3.3.2 Irrigation Water Use

Agriculture has been the Rifle region's primary industry since the late 1920s. Since the Rifle region has been heavily irrigated for decades, both regional and local water quality has been impacted. Springs and other shallow water sources have been created due to irrigation. As irrigation use in the region increased, it became apparent that the Grass Valley Reservoir south of Rifle could not supply all the demand. Initial investigations to expand the irrigation potential for the area were conducted by the Bureau of Reclamation in 1936. Funding for what is referred to as the Silt Project was authorized in 1956. The project was awarded in 1964 and completed in 1967.

Primary achievements of the Silt project were the construction of the Rifle Gap Dam and Reservoir and the Silt Pumping Plant. The Rifle Gap Reservoir is located on Rifle Creek about 5.5 miles north of Rifle at a point where Rifle Creek cuts through the Grand Hogback. Surface runoff from north of Rifle Gap is stored in the reservoir and releases are made to meet downstream irrigation demands. Harvey Gap Reservoir, which previously ran low every year, is now used only after water storage in the Rifle Gap Reservoir has been depleted, thus providing a longer irrigation season.

Prefontaine and Graham Mesas, north and northeast of the New Rifle site, respectively, are some of the most heavily irrigated areas in the Rifle area (Lemon 1995). Irrigation in the Rifle area generally lasts from April through November each year. The quantities of surface runoff collected annually in the Rifle Gap Reservoir and pumpage from the Colorado River at Silt both influence the amount of water which is released from storage at the Harvey Gap Reservoir. Since

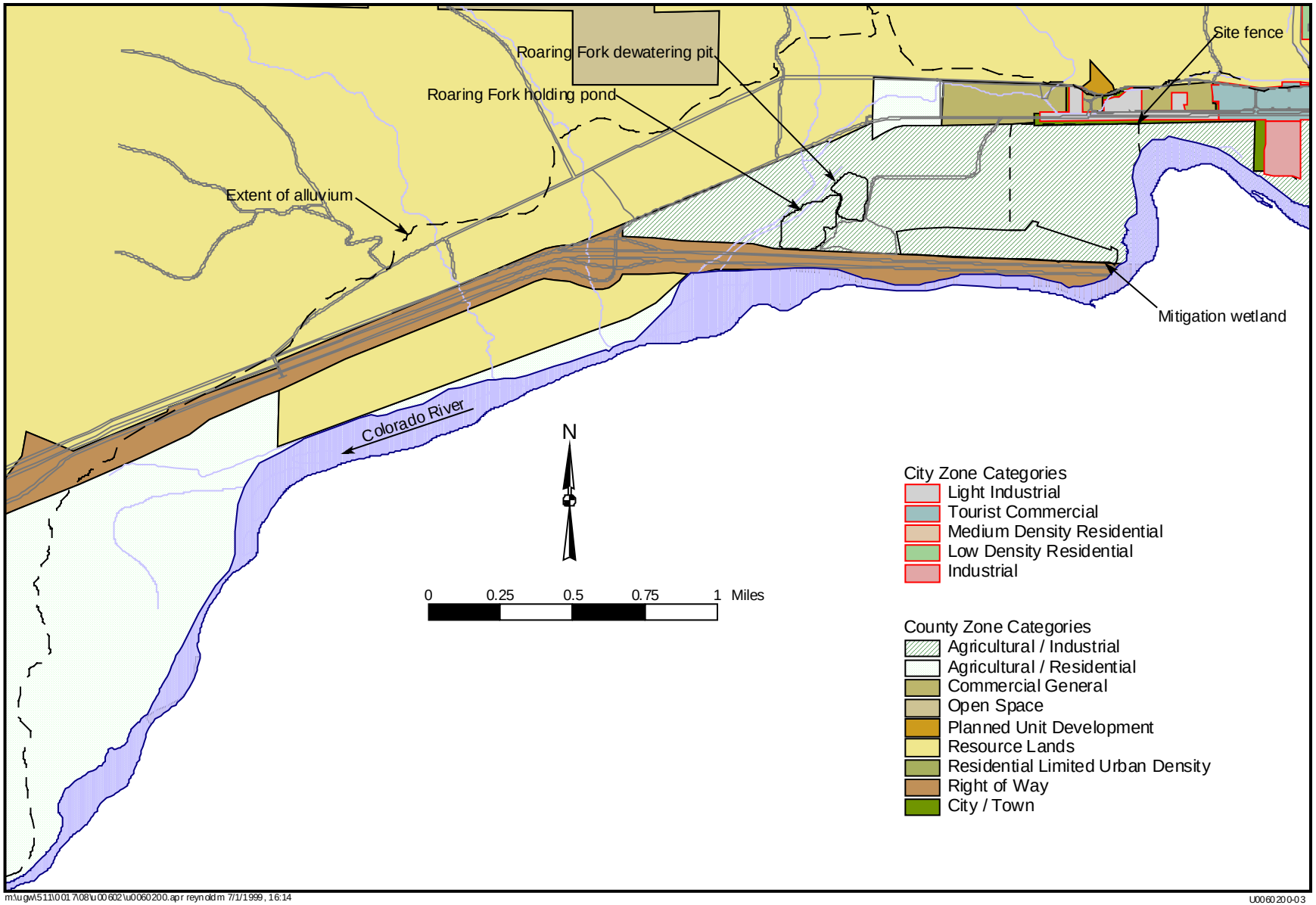

Figure 3–3. Land Use Map for the New Rifle Site and Surrounding Areas

the various irrigation sources each have different water quality and relative regional contributions change on an annual basis as a result of changes in annual precipitation and demand, irrigation in the Rifle area has a significant and variable influence on local ground water flow and water quality.

End of current text

4.0 1998 Field Investigations

The draft SOWP (DOE 1996c) included all previous information collected about the site, presented a proposed compliance strategy, and defined additional data needs that were required to support the most likely compliance strategy. Following stakeholder review and resolution of comments, a field investigation was conducted in 1998 to address the data gaps identified in the draft SOWP. Specifically, additional data was collected to: (1) evaluate if ground water contaminants in the alluvial aquifer will flush naturally in 100 years or less, (2) determine if the alluvial aquifer qualifies for supplemental standards on the basis of widespread ambient contamination, and (3) estimate the incremental risk imposed by mill-related contamination in the alluvial aquifer.

All fieldwork and data quality objectives applied to the data collection activities were performed in accordance with the *Work Plan for Characterization Activities at the UMTRA Project New and Old Rifle Sites* (DOE 1998c). Sequencing the activities to achieve a more logical data collection strategy optimized the field investigations. The field activities were sequenced as follows: (1) collection and analysis of ground water grab samples from temporary well installations, (2) soil sampling, coring, and installation of permanent alluvial and bedrock monitor wells, (3) surveying elevation and location coordinates of wells, (4) ecological and water sampling and analyses, and (5) aquifer testing.

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. Data collection results for the 1998 field investigation and the laboratory analyses are presented in this section. Discussion, interpretation, and integration of the 1998 results with historical information are presented in the subsequent Site Conceptual Model (Section 5.0).

4.1 Ground Water Monitor Well Installations

Temporary and permanent wells were installed to collect water samples for the characterization of the ground water quality and to provide a means to determine hydraulic properties of the alluvial and bedrock aquifers. Installation procedures, construction details, and locations for the wells are described in this section.

4.1.1 Temporary Wells

Nineteen temporary wells were installed at the New Rifle site to collect grab samples of the alluvial ground water for quick turnaround analyses at the Grand Junction Environmental Sciences Laboratory. Quick turnaround for uranium, nitrate, and sulfate concentrations allowed the site conceptual model to be updated as the drilling progressed so that the choice of locations for subsequent temporary and permanent monitor wells could be optimized. Locations where both the temporary and permanent wells were installed are shown in Figure 4–1.

A casing-advance air drilling method was used to advance each borehole through the alluvium to the sampling zone of interest. A 1-inch diameter polyvinyl chloride (PVC) disposable bailer was then inserted through the inside of the drill casing and several bails of ground water from the casing was removed. One to two water samples were then collected to profile the contaminant plume as a function of depth. Advancing the casing to the next deeper zone of interest allowed

collection of multiple samples from the same borehole. Deeper ground water grab samples were collected and analyzed in the same manner as the previous one. Results of the ground water analysis on grab samples collected from the temporary wells are summarized in Table 4–1.

Table 4–1. Analytical Results of Ground Water Grab Samples Collected in 1998 at Temporary Well Locations

Temporary Well	Depth to Bedrock (bgl)[a]	Water Level (bgl)	Saturated Thickness (ft)	Sample Depth (bgl)	Aquifer Zone	Lithology	U (mg/L)	SO_4 (mg/L	NO_3 (mg/L)
168	30	15	15	17	upper	sandy gravel	0.037	506	3.3
				30	lower	sandy gravel	0.009	184	1.9
174	89	88	1	89	lower	sandy gravel	0.054	3,224	10.1
175	18	3	15	7	upper	sandy gravel	0.34	145	3.1
				18	upper	sandy gravel	0.019	166	4
177	54	51	3	54	lower	sandy gravel	0.041	1,198	5.7
178	56	37	19	40	upper	gravel	0.089	2,680	2.7
				56	lower	sand	0.022	654	2.6
180	51	31	20	41	upper	gravelly sand	0.026	2,291	7.9
				51	lower	sandy gravel	0.062	1,928	7.5
181	77	60	17	63	upper	gravel	0.053	2,805	28.6
				77	lower	gravel	0.045	2,499	35.2
182	36	13	23	22	upper	sandy gravel	0.051	1,389	41.8
				36	lower	sandy gravel	0.047	1,838	38.3
183	26	4	22	5	upper	loess	0.049	2,438	2.6
				26	lower	gravelly sand	0.042	1,509	18.5
184	50	34	16	35	upper	sandy gravel	0.051	1,161	38.3
				50	lower	sandy gravel	0.045	1,498	48
185	35	11	24	20	upper	sandy gravel	0.054	2,175	44.4
				35	lower	sandy gravel	0.047	2,072	44
186	27	5	22	11	upper	loess	0.027	824	3.5
				27	lower	sandy gravel	0.012	347	2.2
187	29	6	23	17	upper	gravelly sand	0.06	2,363	22
				29	lower	sandy gravel	0.067	2,689	20.2
188	36	14	22	21	upper	sandy gravel	0.024	3,323	2.2
				36	lower	sandy gravel	0.023	3,786	1.8
189	121	117	4	121	lower	loess	0.016	1,112	2.64
190	29	6	23	6	upper	gravelly sand	0.0681	2,126	3.5
				29	lower	sandy gravel	0.05	3,118	1.8
191	25	9	16	13	upper	sandy gravel	0.0191	2,547	1.76
				25	lower	sandy gravel	0.0542	2,750	6.6
192	21	3	18	9	upper	sandy gravel	0.071	1,946	23.8
				21	lower	sandy gravel	0.069	1,590	11
214	30	Dry	Dry						

[a]below ground level

After the water samples were collected, the temporary casings were removed and the boreholes properly abandoned in accordance with the State of Colorado Water Well Construction Rules (2 CCR 402–2). The following procedures were used for the collection and analyses of the ground water grab samples:

- LQ–11(P), "Standard Practice for Sampling Liquids," *Environmental Procedures Catalog* (GJO 1998).

- ESL Procedure AP(NO_3-1), "Nitrate Analysis," *Environmental Sciences Laboratory Procedures Manual* (MACTEC–ERS 1999).

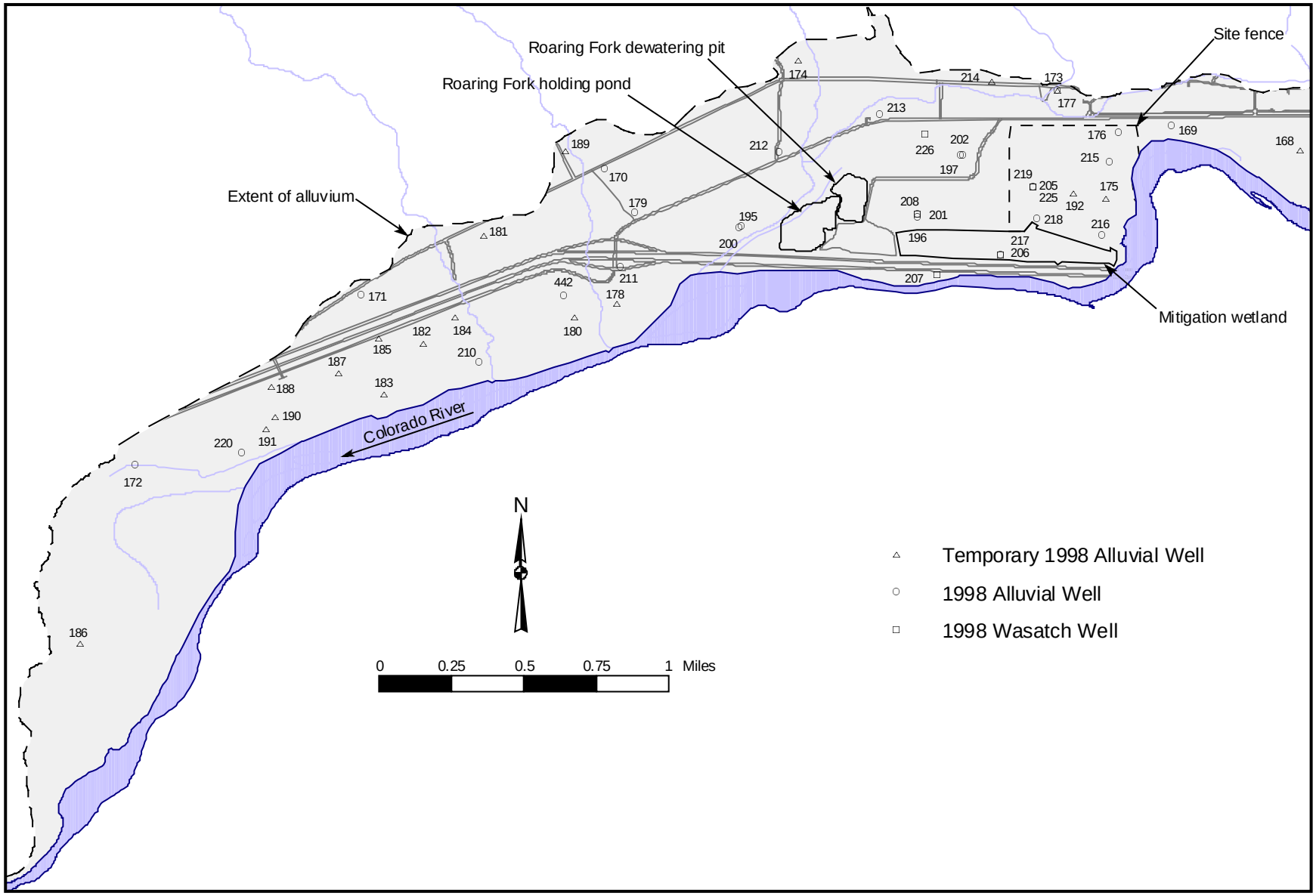

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Figure 4–1. Location of Temporary and Permanent Ground Water Monitor Wells Installed at the New Rifle Site in 1998

- ESL Procedure AP(U-1), "Uranium Analysis," *Environmental Sciences Laboratory Procedures Manual* (MACTEC–ERS 1999).

- ESL Procedure AP(SO_4-1), "Sulfate Analysis," *Environmental Sciences Laboratory Procedures Manual* (MACTEC–ERS 1999).

4.1.2 Permanent Wells

A total of 29 permanent ground water monitor wells were installed in 1998 during the course of the field investigation. One monitor well was installed in a background area east of the site to provide additional information for the characterization of upgradient alluvial ground water that is unaffected by the former vanadium and uranium processing operations. Seven wells were installed on-site, one cross-gradient, and 20 downgradient for the purpose of characterizing the nature and extent of ground water contamination and to determine the hydrologic properties of the aquifers. Twenty-three wells were installed in the alluvial aquifer and six in the bedrock aquifer. These permanent monitor well locations are shown in Figure 4–1.

All permanent alluvial and bedrock wells were completed with 4-inch or 2-inch i.d., flush joint, threaded, PVC casing and wire-wrapped screen (0.020 in slot size). Six of the 23 alluvial monitor wells and all six of the bedrock wells were completed with 4-inch casings so that aquifer testing could be performed. The remaining 17 alluvial wells were all completed with 2-inch casings. The permanent alluvial monitoring wells were screened over the entire saturated thickness in which most cases was nominally 20 ft or less. Two permanent bedrock monitor wells were installed on-site as a paired completion with one shallow well and one deeper well. The shallow well was completed with a 10-ft screen beginning 10 ft below the contact between the alluvium and Wasatch Formation. The deeper bedrock well was completed with a 10-ft screen beginning 30 ft below the contact. The remaining four bedrock wells were installed as shallow completions. Two of these shallow completions were installed adjacent to existing bedrock wells for use as well pairs. The bedrock well pairs with screens at different depths provide information to evaluate the potential impact from the overlaying alluvial plume and the vertical hydraulic gradients in the shallow Wasatch Formation. Drilling and completion techniques to avoid cross-contamination of the Wasatch Formation when drilling the bedrock wells included using a casing-advance drilling method and setting surface casing down into competent bedrock and then drilling through the cement plug into the deeper formation.

All of the permanent wells were constructed with a medium-grained sand pack (10–20 sieve size) placed in the annular space from the bottom of the borehole to 2 ft above the top of the well screen. A fine-grained sand pack (20–40 size) was placed to fill 2 ft of the annular space above the medium-grained sand. Both sand packs consist of clean quartz sand. A 3-ft bentonite seal was placed above the fine-grained sand pack. Enviroplug grout was used to fill the annular space above the bentonite seal to within 3 ft of ground surface. 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. Completion diagrams for the alluvial monitor well installations and the Wasatch wells are presented in Figure 4–2 and Figure 4–3, respectively.

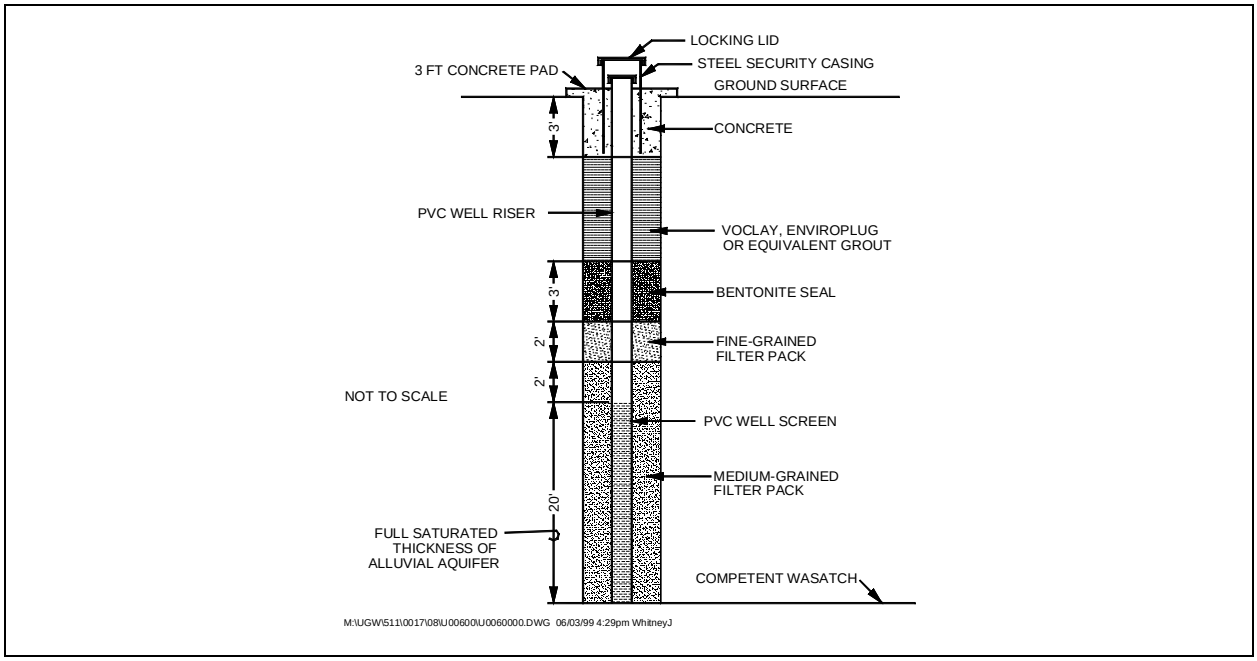


Figure 4–2. Completion Diagram for Wells Installed in the Alluvial Aquifer

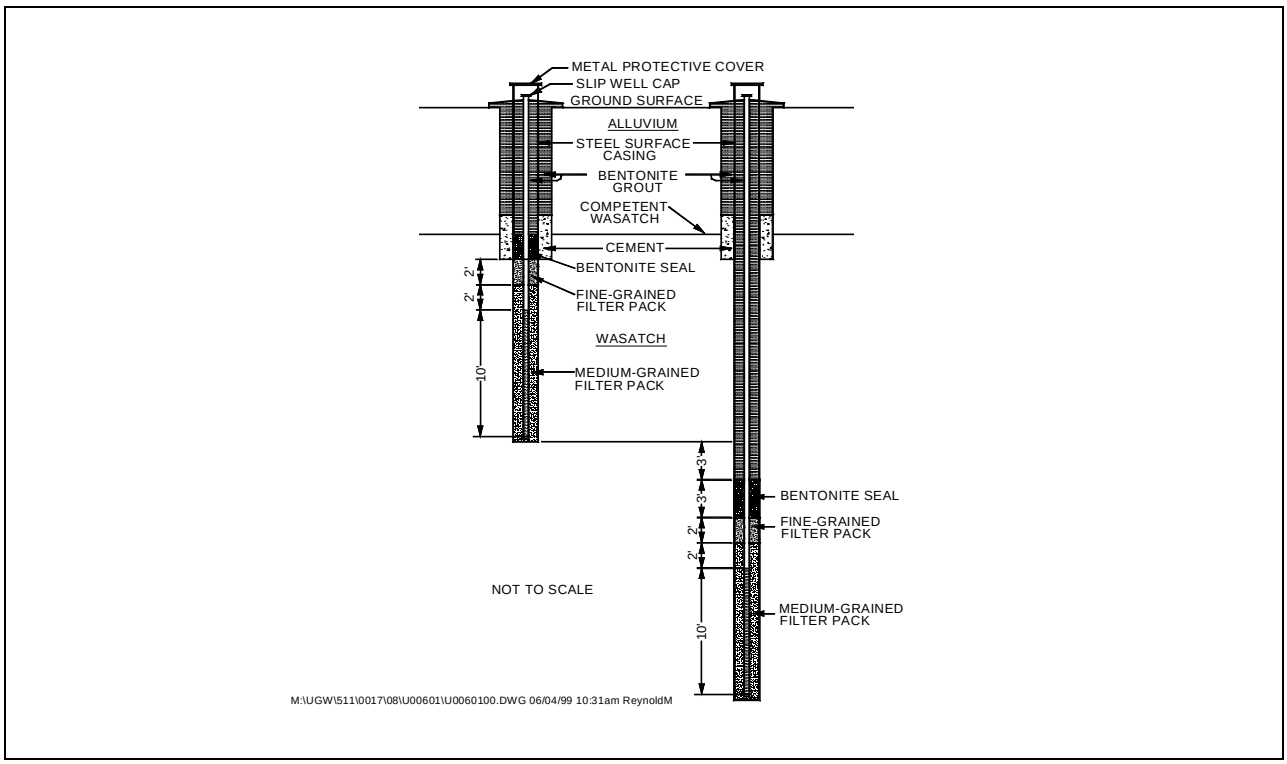


Figure 4–3. Completion Diagram for Shallow and Deep Wells Installed in the Wasatch Formation

The following procedures were used for monitoring 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."

Construction details for the new wells installed during the 1998 field investigation and for existing wells installed prior to the 1998 field investigation are provided in Appendix A and summarized in Tables 4–2 and 4–3. A map showing the locations for the ground water monitor wells installed prior to 1998 at the New Rifle site is provided in Figure 4–4. A comprehensive map showing all the existing monitor wells currently at the New Rifle site is provided in Plate 1 to assist in resolving the many locations.

Table 4–2. Construction Details for Water Wells Installed in 1998 at the New Rifle Site

Location Code	North Coord. (State Plane)	Est Coord. (State Plane)	Ground Elev. (ft)	Borehole Depth (bls)[a]	Borehole Diameter (in)	Top of Casing (ft)	Casing Length (ft)	Casing Diameter (in)	Screen Depth (bls)[a]	Screen Length (ft)	Flow Code[b]	Zone of Completion[c]	Status
0168	624893.93	1351626.88	5284.47	32.00	7.375						U	AL	Abandoned
0169	625316.65	1349269.14	5273.65	19.00	7	5275.47	20.53	2	3.13	15.0	U	AL	Active
0170	624522.41	1338915.04	5332.97	113.00	7	5332.97	112.83	2	92.23	20.0	D	AL	Active
0171	622226.17	1334487.02	5250.68	42.00	7.375	5250.48	41.70	2	26.33	15.0	D	AL	Active
0172	619125.31	1330342.23	5226.47	35.00	7.375	5229.45	35.56	2	6.98	25.0	D	AL	Active
0173	625962.55	1347188.96	5311.54	57.60	7.375	5313.94	59.77	2	46.80	10.0	C	AL	Active
0174	626503.18	1342447.33	5327.54	97.00	7.375						D	AL	Abandoned
0175	624005.1	1348055.77	5264.33	19.00	7.375						O	AL	Abandoned
0176	625185.73	1348311.73	5286.6	38.50	7.375	5286.6	37.80	2	17.23	20.0	O	AL	Active
0177	625952.55	1347188.96		55.00	7						C	AL	Abandoned
0178	622083.25	1339138.82	5268.05	57.00	7						D	AL	Abandoned
0179	623712.48	1339472.27	5302.56	83.50	7	5302.56	80.74	2	60.14	20.0	D	AL	Active
0180	621809.66	1338385.94	5266.3	52.00	7.375						D	AL	Abandoned
0181	623305.93	1336709.32	5293.89	78.00	7						D	AL	Abandoned
0182	621348.38	1335623.95	5243.36	38.00	7.375						D	AL	Abandoned
0183	620404.72	1334911.65	5231.43	27.00	7.375						D	AL	Abandoned
0184	621821.54	1336194.89	5259.72	53.00	7.375						D	AL	Abandoned
0185	621448.52	1334788.5	5240.27	37.00	7.375						D	AL	Abandoned
0186	615860.85	1329351.69	5212.24	28.00	7.375						D	AL	Abandoned
0187	620796.62	1334074.69	5233.07	33.00	7.375						D	AL	Abandoned
0188	620562.58	1332827.26	5238.17	37.00	7.375						D	AL	Abandoned
0189	624856.11	1338200.69	5356.07	123.00	7.375						D	AL	Abandoned
0190	620005.52	1332921.88	5228.3	32.00	7.375						D	AL	Abandoned
0191	619766.72	1332742.56	5227.34	28.00	7						D	AL	Abandoned
0192	624070.79	1347477.22	5263.69	22.00	7.375						O	AL	Abandoned
0195	623469.99	1341416.01	5250.38	26.50	9.343	5253.1	28.34	4	5.29	20.0	D	AL	Active
0196	623654.13	1344643.5	5258.25	22.00	9.343	5260.99	24.21	4	6.14	15.0	D	AL	Active
0197	624776.77	1345465.37	5258.67	23.50	9.343	5261.88	25.33	4	1.79	20.0	D	AL	Active
0200	623443.5	1341375.75	5250.68	28.50	9.343	5252.68	30.33	4	8.00	20.0	D	AL	Active
0201	623703.91	1344647.96	5258.27	23.00	9.343	5261.07	25.48	4	7.35	15.0	D	AL	Active
0202	624775.51	1345413.53	5257.96	25.50	9.343	5260.43	27.76	4	4.96	20.0	D	AL	Active
0205	624181.3	1346732.27	5263.87	60.05	7.375	5267.07	63.23	4	49.70	10.0	O	WS	Active
0206	622956.94	1346148.3	5255.4	38.00	7.375	5258.2	40.33	4	27.20	10.0	D	WS	Active
0207	622590.12	1344989.18	5256.93	43.33	7.375	5256.93	43.33	4	33.00	10.0	D	WS	Active
0208	623704.23	1344638.17	5258.78	38.00	7.375	5260.78	40.00	4	27.70	10.0	D	WS	Active
0210	620982	1336626.79	5246.13	38.00	7.375	5249.13	40.39	2	16.82	20.0	D	AL	Active
0211	622736.96	1339190.84	5286.97	73.00	7	5286.97	72.25	2	51.65	20.0	D	AL	Active
0212	624838.51	1342088.94	5285.62	57.00	7.375	5285.62	57.00	2	36.40	20.0	D	AL	Active
0213	625507.49	1343944.68	5266.4	32.00	7.375	5265.83	31.20	2	6.20	25.0	D	AL	Active
0214	626122.25	1346000.06	5311.9	32.00	7						C		Abandoned
0215	624655.44	1348154.2	5268.52	22.50	7.375	5271.42	25.31	2	6.84	15.0	O	AL	Active
0216	623300.84	1348007.04	5262.13	21.50	7.375	5265.41	24.35	2	5.50	15.0	O	AL	Active
0217	622965.11	1346146.38	5255.28	23.00	7	5256.98	24.68	2	7.40	15.0	D	AL	Active
0218	623626.81	1346817.6	5262.19	22.15	7.375	5265.24	25.20	2	6.58	15.0	O	AL	Active
0219	624194.4	1346724.7	5263.78	27.50	7.375	5266.58	29.90	2	6.53	20.0	O	AL	Active
0220	619335.1	1332293.15	5225.25	33.00	7.375	5228.6	33.84	2	4.89	25.0	D	AL	Active
0225	624180.49	1346722.07	5263.97	40.00	7.375	5266.97	43.00	4	29.77	10.0	O	WS	Active
0226	625148.64	1344770.96	5259.58	41.50	7.375	5262.08	43.83	4	31.00	10.0	D	WS	Active
0442	622216.34	1338182.58				5279.03					D	AL	Private/Active

[a]bls = below land surface
[b]C = crossgradient D = downgradient N = unknown O = on-site U = upgradient
[c]AL = alluvium WS = Wasatch formation-undifferentiated

Table 4–3. Construction Details for Water Wells Installed Prior to 1998 at the New Rifle Site

Location Code	North Coord. (State Plane)	East Coord. (State Plane)	Ground Elev. (ft)	Borehole Depth (bls)[a]	Borehole Diameter (in.)	Top of Casing Elev. (ft)	Casing Length (ft)	Casing Diameter (in)	Screen Depth (bls)[a]	Screen Length (ft)	Flow Code[b]	Zone of Completion[c]	Status
0581	623212.6	1346316	5264.7	26.50	6.625	5265.9	24.10	4	17.90	5.0	O	AL	Abandoned
0582	623131.6	1346324.3	5264.5	16.50	6.625	5265.72	12.52	4	7.30	4.0	O	AL	Abandoned
0583	623196.4	1346314.6	5264.4	17.00	6.625	5265.8	17.90	4	12.50	4.0	O	AL	Abandoned
0584	623470.1	1346520.3	5314.9	75.00	6.625	5316.61	74.81	4	69.60	3.5	O	AL	Abandoned
0585	625132.8	1348610.8	5275.4	28.00	6.625	5275.39	24.99	4	8.00	17.0	O	AL	Abandoned
0586	623434.7	1346524.3	5315.1	61.00	6.625	5316.61	62.21	4	57.70	3.0	O	AL	Abandoned
0587	623455	1346526	5315.04	66.00	6.625	5315.94	66.90	4	63.00	3.0	O	AL	Abandoned
0588	623701.7	1347475.3	5265.7	25.50	6.625	5265.73	25.53	4	5.50	20.0	O	AL	Abandoned
0589	623257.37	1344251.04	5256.54	22.00	6.625	5257.86	23.32	4	7.00	15.0	D	AL	Active
0590	623244.7	1345383.69	5254.31	26.91	6.625	5256.37	21.27	4	5.21	14.0	D	AL	Active
0591	624928.1	1348693	5272.8	13.50	6	5275.01	15.51	2	6.30	5.0	O	AL	Abandoned
0592	624921.8	1348684.1	5272.6	22.00	6	5275.15	24.35	2	14.80	5.0	O	AL	Abandoned
0593	624253.7	1347382.5	5276.6	22.50	6	5278.18	19.48	2	10.90	5.0	O	AL	Abandoned
0594	624261.4	1347389.9	5276.4	26.50	6	5278.13	25.23	2	16.50	5.0	O	AL	Abandoned
0595	623163.7	1346365.5	5264.7	23.00	10	5266.77	24.07	6	5.00	15.0	O	AL	Abandoned
0596	623070.47	1343250.2	5255.05	13.50	6	5257.11	15.56	2	6.50	5.0	D	AL	Active
0598	624312.8	1344582.67	5255.2	12.00	6	5257.09	13.39	2	4.50	5.0	D	AL	Active
0599	624314.81	1344572.79	5255.5	17.00	6	5257.1	18.60	2	10.00	5.0	D	AL	Active
0600	622993.3	1345807.2	5255.75	12.65	6	5255.75	12.65	2	5.65	5.0	D	AL	Active
0601	623007	1345824.5	5260.7	12.00	6	5263.02	14.32	2	5.00	5.0	D	AL	Abandoned
0602	623428.41	1341388.84	5250.99	12.00	6	5253.48	13.49	2	4.00	5.0	D	AL	Active
0603	623420.95	1341394.73	5250.85	24.00	6	5253.79	25.94	2	16.00	5.0	D	AL	Active
0604	621822.39	1339038.75	5260.03	12.00	6	5261.64	13.61	2	5.00	5.0	D	AL	Active
0605	621822.08	1339049.53	5260.09	25.00	6	5261.8	26.71	2	18.00	5.0	D	AL	Active
0608	623363.87	1339291.87	5300.56	30.00	6	5302.07	31.51	2	8.00	20.0	D	AL	Active
0609	624827.2	1343083.9	5257.99	23.00	6	5260.19	25.20	2	6.00	15.0	D	AL	Active
0610	625242.5	1346191.5	5278.5	47.50	6	5280.29	36.79	2	3.00	30.0	D	AL	Abandoned
0611	623223.5	1346324.29	5257.38	65.58	6	5259.56	67.97	2	15.37	15.0	O	WS	Active
0612	623154.8	1346340.3	5264	71.00	6	5265.61	71.11	6	52.87	15.0	O	WS	Abandoned
0613	623143	1346317.7	5257.77	65.37	10	5264.71	73.00	6	15.43	43.0	O	WS	Abandoned
0614	623162.7	1346333.5	5257.45	36.35	6	5258.16	37.48	2	18.02	2.0	O	WS	Abandoned
0615	625071.4	1346756.2	5277.2	32.00	6	5278.62	30.42	2	7.00	20.0	O	AL	Abandoned
0616	622918.3	1346603.1	5264.7	21.00	6	5266.01	22.31	2	9.00	10.0	O	AL	Abandoned
0617	621127	1334656		50.00	9		50.00	7	40.00	10.0	D	AL	Private/Active
0618	623073.25	1343239.48	5255.24	22.00	6	5256.97	23.73	2	15.00	5.0	D	AL	Active
0619	623207.6	1346298.5	5263.6	15.00	6	5265.84	15.64	2	6.40	5.0	O	AL	Abandoned
0620	619811.71	1332926.38	5227.8	10.70	0	5231.22	12.17	2	6.70	4.0	D	AL	Active
0621	621828.04	1338969.97	5257.6	120.00	8	5259.68	122.08	4	98.00	20.0	D	WS	Active
0622	621851.17	1338971.99	5258.54	60.00	6	5260.23	61.69	2	53.00	5.0	D	WS	Active
0623	622956.6	1346086.2	5261.44	96.00	8	5262.92	97.48	4	45.00	20.0	D	WS	Abandoned
0624	622957.64	1346138.76	5255.93	42.45	6	5257.92	44.44	2	22.45	10.0	D	WS	Active
0625	623987.34	1348111.1	5264.37	102.00	8	5266.79	104.42	4	90.00	10.0	O	WS	Active
0626	623971.4	1348165.79	5264.7	78.00	6	5267.42	80.72	2	66.00	10.0	O	WS	Active
0627	622980.14	1346354.54	5255.51	84.99	8	5257.48	86.96	4	58.99	20.0	O	WS	Active
0628	622921.68	1346457.26	5260.55	111.89	8	5263.97	114.31	4	88.89	20.0	O	WS	Active
0629	623022.17	1346426.03	5256.1	49.35	6	5257.34	50.59	2	37.35	10.0	O	WS	Active
0630	623658.97	1345810.57	5254.76	88.12	8	5256.21	84.57	4	61.12	20.0	D	WS	Active
0631	623688.48	1345811.21	5253.39	49.49	6	5254.5	50.60	2	37.49	10.0	D	WS	Active
0632	625110.58	1346363.02	5276.24	102.25	8	5277.84	97.85	4	84.25	10.0	O	WS	Active
0633	625056.36	1346366.63	5275.85	72.99	6	5278.64	75.78	2	60.99	10.0	O	WS	Active
0634	624017.68	1348123.71	5264.94	51.23	6	5267.4	53.69	2	39.23	10.0	O	WS	Active
0635	622500.75	1344936.24	5253.5	20.10	7.875	5253.12	16.62	4	12.00	5.0	D	AL	Active
0636	622582.36	1344996.84	5256.8	19.66	7.875	5256.24	18.54	4	14.10	5.0	D	AL	Active
0640	626269.8	1347087.1	5335.02	160.00	8	5335.9	153.88	4	131.00	20.0	C	WS	Active
0641	626237.4	1347086.8	5333.75	86.50	6	5334.63	87.38	2	70.50	10.0	C	WS	Active
0642	623849.1	1344904.17	5257.42	105.00	8	5258.87	103.45	4	72.00	20.0	D	WS	Active
0643	623868.59	1344874.65	5257.12	71.76	6	5258.15	72.47	2	59.44	10.0	D	WS	Active
0644	622173	1348429.8	5268.35	100.00	8	5269.85	101.50	4	85.00	10.0	C	WS	Active
0645	622177.3	1348471.4	5268.78	72.00	6	5270.7	73.92	2	60.00	10.0	C	WS	Active
0646	625231.7	1350923.3	5278.18	150.00	8	5279.96	151.78	4	120.00	20.0	U	WS	Abandoned
0647	625190	1350922.5	5277.94	78.00	8	5279.44	79.50	4	58.00	15.0	U	WS	Abandoned
0650	623451.83	1343197.74	5252.84	102.00	8	5254.1	103.26	4	82.00	10.0	D	WS	Active
0651	623495.49	1343197.26	5253.16	65.00	6	5254.6	66.44	2	53.00	10.0	D	WS	Active
0655	625128.36	1347150.57	5273.92	17.00	2	5276.91	19.50	2	6.50	10.0	O	AL	Active
0656	624879.91	1346651.44	5268.84	12.60	2	5271.29	14.50	2	2.00	10.0	O	AL	Active
0657	624989.77	1347369.13	5271.33	18.30	2	5274.13	20.80	2	8.00	10.0	O	AL	Active
0658	624379.97	1347302.91	5264.79	5.40	2	5265.91	6.60	2	0.50	5.0	O	AL	Active
0659	623422.58	1346432.69	5259.24	10.50	2	5261.33	12.60	2	0.50	10.0	O	AL	Active
0663	625058.08	1346572.71	5271.41	15.80	7.625	5273.38	17.77	2	8.50	7.0	O	AL	Active
0664	624664.05	1346562.68	5268.01	15.00	7.625	5270.17	17.16	2	7.70	7.0	O	AL	Active
0668	624484.15	1345914.17	5262.92	10.80	7.625	5265.19	13.07	2	5.50	5.0	D	AL	Active
0669	624268.75	1346554.96	5264.33	15.50	7.625	5266.56	13.23	2	4.00	6.6	O	AL	Active
0670	624808.1	1346912.14	5268.72	12.60	7.625	5270.94	14.82	2	5.20	7.0	O	AL	Active
0675	624497.06	1345421.1	5259.96	12.00	7.625	5262.21	14.25	2	5.00	6.6	D	AL	Active
0677	624796.84	1345414.19	5258.31	9.00	7.625	5260.15	10.84	2	4.00	4.6	D	AL	Active
0678	624930.49	1346730.45	5270.02	46.95	8	5272.26	49.19	2	34.65	12.0	O	WS	Active
0679	625303.37	1347464.15	5287.74	27.63	8	5290.07	29.96	2	22.33	5.0	O	AL	Active

[a]bls = below land surface
[b]C = crossgradient D = downgradient N = unknown O = on-site U = upgradient
[c]AL = alluvium WS = Wasatch formation-undifferentiated

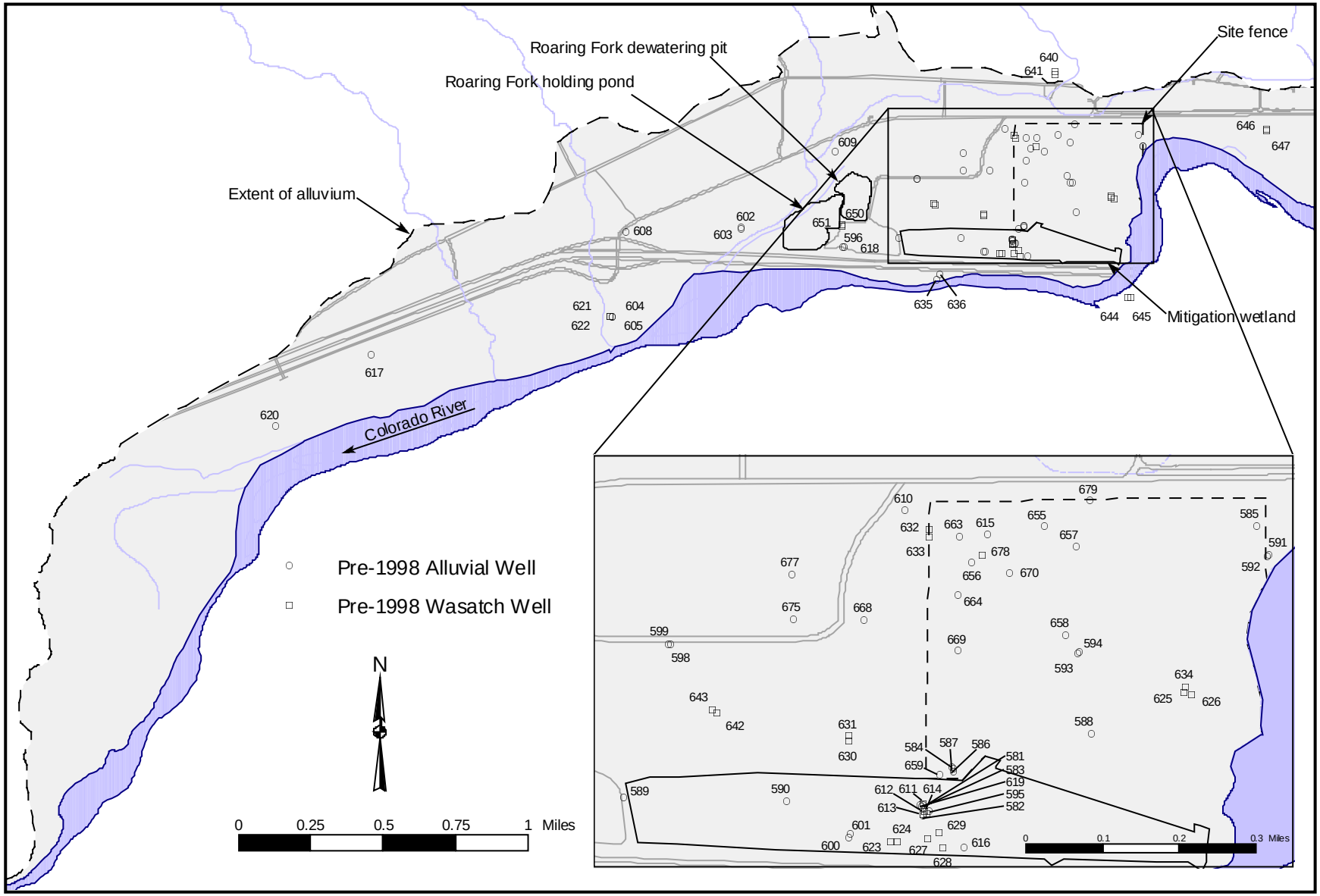


Figure 4–4. Location of Ground Water Monitor Wells Installed at the New Rifle Site Prior to 1998

4.2 Soil, Sediment, and Rock Sampling

Soil, sediment, and rock samples were collected during the installation of the temporary and permanent ground water monitor wells for lithologic logging and chemical analysis. Lithologic logs were prepared for all monitor well boreholes to support development of the site hydrogeologic model. Sediment, soils, and rock samples were collected for chemical analysis to determine distribution coefficients (K_d) and mobile fractions of site-related constituents in the subpile soils to aid in characterizing subsurface contaminant transport. Descriptions of the methods used to collect grab, split-spoon, and rock core samples are provided in this section. Procedures used for the lithologic logging and chemical analysis of the samples are provided in Sections 4.3, 4.4, and 4.5.

A truck-mounted Mobile B–80 drilling rig equipped with a downhole air hammer was used to advance casing through the alluvial aquifer to bedrock. During the drilling the air was continuously returned to the surface inside the casing where it was diverted to a cyclone sampler. Grab samples of the drill cutting entrained in the return air were collected at the cyclone's discharge every 5 ft of depth drilled.

Split-barrel sampling was conducted with the truck-mounted casing-advance drilling rig centered over the sample location. After advancing the casing to the desired sampling depth a 3-inch o.d. by 24-inch-long split-barrel sampler was lowered inside the casing to the top of the sample interval. A 140-pound drop hammer was then used to drive the split-barrel sampler the required 2 ft or until penetration was less than 6 inches per 50 blows. The barrel was then removed from the borehole, separated from the drive-rod assembly, and laid flat on an uncontaminated surface, where the head and drive shoe were removed. One-half of the split barrel was removed to expose the sample. The uppermost portion of sample in the split barrel was inspected for slough and the slough discarded, if present. The site geologist logged the remaining representative sample.

Continuous rock core samples of the Wasatch Formation were collected using a nominal 5-ft long, double tube, swivel-type, NX diamond core barrel and wireline system. Clean water was used as the circulation medium. State-of-the industry coring practices were used to effect the highest core recovery possible. Recovered core was washed and then placed in boxes within the longitudinal separators, from left to right, as a book would be read; that is, core was placed starting with the shallowest portion of the hole in the upper left hand 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. The site geologist described all cores.

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

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

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

- SL–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."

4.3 Lithologic Logging

Lithologic logging of the alluvial material was performed primarily on drill cutting samples that were collected from the cyclone every 5 ft of drilled depth during the course of the monitor well installations. Split-barrel sampling for lithologic purposes was attempted once every 5 ft of drilled depth. However, in most cases a large amount of undisturbed sample could not be recovered due to the difficulty in collecting unconsolidated silt, sand, and cobble gravel with a spoon type sampling device.

Lithologic logging of the Wasatch Formation was based primarily on continuous diamond core samples collected during the installation of the bedrock monitor wells. Approximately 34 ft of competent rock core was collected at the lower Wasatch completion RFN–205 and approximately 17 ft of rock core was collected at the upper Wasatch completions RFN–207 and –226. All cores were logged for lithology. Split-barrel sampling was also attempted at the bottom of each temporary and permanent alluvial well boring that was not cored to verify the bedrock contact. In most cases, an adequate amount of Wasatch material was recovered by the split-barrel method to confirm that bedrock was reached.

Lithologic descriptions of the drill cuttings, split-barrel samples, and rock core recorded by the site geologist 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 1998):

- 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)."

4.4 Distribution Coefficient (K_d) Analysis

The K_d is a bulk parameter that has been used with some success to describe the retardation of contamination in an aquifer system. Laboratory measurements to determine the K_d for selected analytes were performed on alluvial and Wasatch material to support computer-modeling efforts in characterizing subsurface contaminant transport at the New Rifle site. Details regarding the laboratory procedure used to determine the K_d values are presented in the Calculation Set U0058100. A summary of the results is provided in the following sections.

4.4.1 Method of Solution

Laboratory analyses of the K_d were performed according to American Society of Testing Materials (ASTM) procedure D 4646–87 (ASTM 1987) for selected regulated and commonly retarded site-related contaminants of potential concern (COPCs); arsenic (As), molybdenum (Mo), selenium (Se), uranium (U), and vanadium (V). Essentially, the ASTM procedure involves placing a sample representative of a site (e.g., soil, sediments, cuttings, core) 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 then 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

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

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 K_d, the greater the retardation of a contaminant in ground water.

The ASTM 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 K_d for contaminants with higher adsorptive properties, as the adsorptive ability of the sample would be reduced by contaminants already present.

4.4.1.1 Sample Selection

Alluvial and Wasatch samples collected during the installation of monitor wells were selected for K_d testing. Sample collection procedures are described in Section 4.2. Sample locations are shown in Figure 4–1.

Some of the alluvial samples had very limited recovery and sample size was not adequate to perform the analyses because they consisted mainly of very coarse cobbles and other materials for which K_d testing is not appropriate. However, ten alluvial samples collected at four downgradient locations, RFN–171, –189, –210, and –214, were adequate in volume for K_d testing. Sample 171 was collected near the top of the saturated alluvium. Sample 189 was collected near the base of the saturated alluvium. Samples at location 210 were selected from both shallow and deep portions of the saturated alluvium. Duplicate samples were collected from two zones near the water table and from one zone near the base of the saturated alluvium. Two samples were collected from the upper alluvial material at location 214. While these samples are located downgradient of the site, they appear to be from locations outside the contaminant plume and unaffected by site contamination.

Three Wasatch samples were selected from two monitor well locations for K_d analysis; one on-site core hole (RFN–205) and one off-site core hole (RFN–208). Two samples were collected from a shallow depth and one sample was collected from a deeper depth. The shallow samples (208 and 205B) were obtained 6 to 7 ft below the top of the Wasatch Formation and consist mostly of fine-grained sandstone. Abundant organic matter was noted in sample 208 and minor

claystone lenses were present in sample 205B. The deeper-depth sample (205A) was obtained 24 ft below the top of the Wasatch Formation and consists of a fine-grained sandstone with frequent claystone lenses.

4.4.1.2 Sample Preparation

All the samples were air dried at room temperature and sieved to less than 10 mesh (2 millimeters [mm]). A riffle splitter is used to separate a sample for oven drying at 105 °C to determine moisture content. The difference between the air-dried and oven-dried weights was always less than 2 percent and usually 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 at the New Rifle site. Based on ground water analysis for well 598 from sampling conducted in April and September 1997 (SEE_UMTRA data base), the composition of the synthetic ground water was prepared as follows (in mg/L): Na = 584, K = 7.2, Ca = 328, Mg = 69.3, SO_4 = 1,626, Cl = 203, NO_3 = 639, and C (inorganic) = 121. Contaminants were then added to obtain the following target concentrations (mg/L): As = 0.2, Mo = 5.0, Se = 0.3, U = 0.5, and V = 10. The pH was adjusted to about 6.5 and the measured alkalinity was about 300 mg/L as $CaCO_3$.

4.4.1.3 Sample Analysis

Analysis of the synthetic ground water solution (Syn–1 and –2) was performed to verify the target concentrations were achieved. These results are reported in Table 4–4. Actual As concentrations are slightly lower than the target value of 0.2 mg/L while Mo, Se, U, and V are consistent with their target concentrations of 5.0, 0.3, 0.5, and 10 mg/L, respectively.

Table 4–4. Laboratory Analytical Results for Synthetic Ground Water Solution and Sample Leachate

Sample ID	Sample Description	Solution Volume (mL)	Sample Volume (g)	Concentration (mg/L)				
				As	Mo	Se	U	V
Syn–1	Synthetic ground water	100	NA	0.136	4.51	0.352	0.573	9.2
Syn–2	Synthetic ground water	100	NA	0.147	4.6	0.289	0.47	9.56
171	Alluvium leachate	100	5.0	0.0921	4.58	0.237	0.452	6.9
189	Alluvium leachate	100	5.0	0.0706	4.45	0.266	0.547	5.38
210–A–1	Alluvium leachate	100	5.0	0.106	4.68	0.248	0.471	7.35
210–B–1	Duplicate	100	5.0	0.106	4.58	0.242	0.473	7.29
210–A–2	Alluvium leachate	100	5.0	0.12	4.33	0.26	0.439	8.05
210–B–2	Duplicate	100	5.0	0.122	4.56	0.256	0.467	8.13
210–B–3	Alluvium leachate	100	5.0	0.109	4.61	0.245	0.46	7.26
210–A–3	Duplicate	100	5.0	0.0988	4.38	0.245	0.448	6.75
214	Alluvium leachate	100	5.0	0.064	4.35	0.28	0.517	3.93
214–A	Alluvium leachate	100	5.0	0.0758	4.55	0.207	0.441	4.54
208	Wasatch leachate	100	5.0	0.0245	4.28	0.14	0.482	2.38
205–B	Wasatch leachate	100	5.0	0.0802	4.37	0.336	0.574	7.14
205–A	Wasatch leachate	100	5.0	0.079	4.42	0.264	0.562	5.21

Approximately 5 grams of each sample were measured and placed in 125-milliliter (mL) Nalgene 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 at 3,000 rpm and filtered through a 0.45 micrometer (μm) filter. The resulting leachate samples were preserved with 1 percent nitric acid and submitted to the Grand Junction Office (GJO) Analytical Chemistry Laboratory for analysis of As, Mo, Se, U, and V. Analytical results are reported in Table 4–4.

4.4.1.4 K_d Calculation

K_ds are calculated using the analytical data summarized in Table 4–4 and the following equation:

$$K_d = \frac{(A-B)V}{(M_s)B} \qquad \text{where}$$

A = initial concentration (mg/L) of the COPCs in the synthetic ground water (defined as the mean concentration obtained for the synthetic ground water samples Syn–1 and –2)
B = final concentration of the COPCs in the leachate, after 24 hours in contact with the sediment sample (mg/L)
V = volume of solution (mL)
M_s = mass of sediment sample (grams)
K_d = distribution coefficient (milliliters per gram [mL/g])

Results of the calculations are presented in Table 4–5. Note that the K_d in Table 4–5 is the same as the Rd in the ASTM procedure; this value only represents a true K_d if equilibrium conditions were attained during the test period.

Table 4–5. Measured K_d Values for Alluvium and Wasatch Samples at the New Rifle Site

Monitor Well	Sample ID	Sample Depth (ft below surface)	Sample Description	K_d (mL/g)				
				As	Mo	Se	U	V
171	171	23–28	Alluvium at water table	10.7	-0.1	7.0	3.1	7.2
189	189	121	Alluvium at base of saturated zone	20.1	0.5	4.1	-0.9	14.9
210	210–A–1	17	Alluvium at water table	6.7	-0.5	5.8	2.1	5.5
	210–B–1	17	Duplicate sample	6.7	-0.1	6.5	2.1	5.7
	210–A–2	15–16	Alluvium at water table	3.6	1.0	4.7	3.8	3.3
	210–B–2	15–16	Duplicate sample	3.2	0.0	5.0	2.3	3.1
	210–B–3	27	Alluvium at base of saturated zone	6.0	-0.2	6.2	2.7	5.8
	210–A–3	27	Duplicate sample	8.6	0.8	6.2	3.3	7.8
214	214	7–8.2	Near top of alluvium – dry	24.2	0.9	2.9	0.2	27.7
	214–A	12.5	Near top of alluvium – dry	17.3	0.0	11.0	3.7	21.3
208	208	27	7-ft below Wasatch contract – sandstone	95.5	1.3	25.8	1.6	58.8
205	205–B	34.5	8-ft below Wasatch contract – sandstone	15.3	0.8	-0.9	-1.8	6.3
	205–A	50	24-ft below Wasatch contract – sandstone	15.8	0.6	4.3	-1.4	16.0

Measured K_d values obtained for the alluvial aquifer samples were adjusted based on aquifer grain size analysis. As part of an adsorption study performed by Jacobs Engineering Group Inc. (Jacobs 1993), over 400 pounds of material from each of seven test pits excavated in the alluvial aquifer (totaling over a ton of material) was sorted and analyzed for grain size distribution. The study found that on average, only 38 percent of the aquifer material was contained in the <2 mm size fraction and that the majority was gravel size. Therefore, the measured K_d values for the alluvial aquifer, which were performed on the <2 mm size fraction, were adjusted by multiplying by 0.38. This assumes that the gravel-sized materials are insignificant in terms of contaminant adsorption. The adjusted values are reported in Table 4–6.

Table 4–6. Results of Alluvial K_d Values Adjusted for Grain Size Distribution

Monitor Well	Sample ID	Sample Depth (ft below surface)	Sample Description	K_d (mL/g)				
				As	Mo	Se	U	V
171	171	23–28	Alluvium at water table	4.1	0.0	2.7	1.2	2.7
189	189	121	Alluvium at base of saturated zone	7.6	0.2	1.6	-0.3	5.7
210	210–A–1	17	Alluvium at water table	2.5	-0.2	2.2	0.8	2.1
	210–B–1	17	Duplicate sample	2.5	0.0	2.5	0.8	2.2
	210–A–2	15–16	Alluvium at water table	1.4	0.4	1.8	1.4	1.3
	210–B–2	15–16	Duplicate sample	1.2	0.0	1.9	0.9	1.2
	210–B–3	27	Alluvium at base of saturated zone	2.3	-0.1	2.4	1.0	2.2
	210–A–3	27	Duplicate sample	3.3	0.3	2.4	1.3	3.0
Mean 210				2.2	0.1	2.2	1.0	2.0
214	214	7–8.2	Near top of alluvium – dry	9.2	0.3	1.1	0.1	10.5
	214–A	12.5	Near top of alluvium – dry	6.6	0.0	4.2	1.4	8.1
Mean 214				7.9	0.2	2.6	0.7	9.3

4.5 Subpile Soil Analysis

All radioactive tailings and material were removed from the New Rifle site in 1996 as part of the surface remediation program. No radioactive materials exceeding 15 picocuries per gram (pCi/g) radium-226 were left. However, the potential exists for nonradionuclide contaminants to have seeped into the soils below the depth of remediation. Contaminated soils could contaminate infiltrating water as it passes through them and prolong the ground water cleanup effort. Details regarding the laboratory procedure used to evaluate the potential for residual contaminants in the subpile soils are presented in Calculation Sets U0058100 and U0045300. A summary of the results is provided in the following sections.

4.5.1 Subpile Soil Sample Selection

Ten samples from six boreholes were collected as described in Section 4.2. Two of the boreholes (RFN–218 and –225) are located in the former footprint of the New Rifle tailings pile, one borehole (RFN–216) is located near the center of the former evaporation pond, and three boreholes (RFO–292, –658, and –659) are located in background areas upgradient of the site, approximately 1 mile east of the town of Rifle. The on-site and upgradient background locations are shown in Figure 4–5.

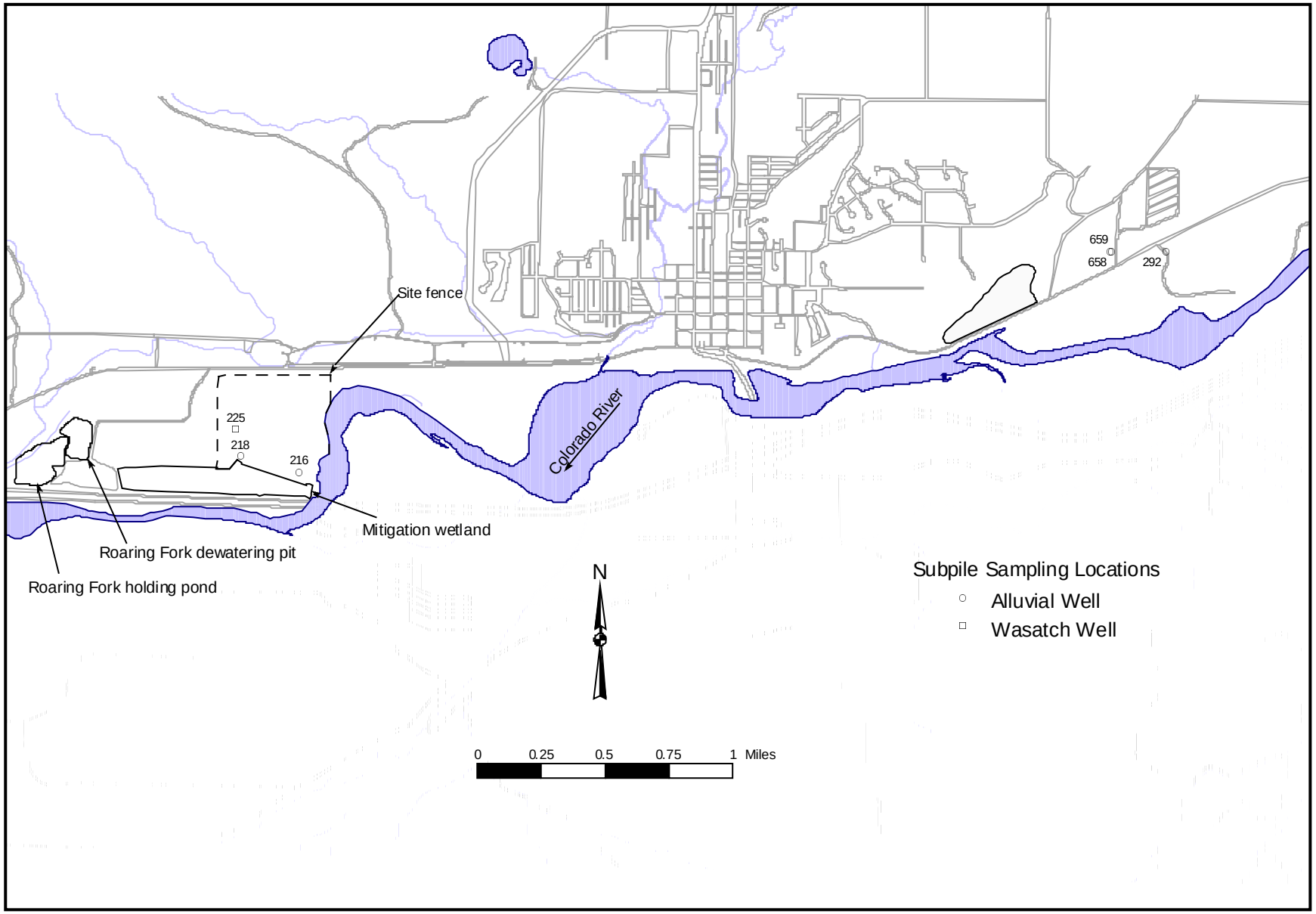


Figure 4–5. Subpile Sample Locations

4.5.2 Subpile Soil Sample Preparation and Extraction

Chemical extractions were used to determine the potential mobility of contaminants. Each sample was extracted by using two separate lixiviants; a synthetic ground water solution and a 5 percent nitric acid solution. Extractions were performed sequentially on the same starting material to avoid variation due to sample heterogeneity. Each extraction was harsher than the preceding one.

Each chemical extraction was related to a scenario that could cause contaminant release at the site. Synthetic ground water was used first to simulate the water-table rise that could cause ground water to contact contaminated soils. The synthetic solution was prepared based on ground water analysis for well RFN–598 from two sampling events in 1997 (SEE_UMTRA data base). The composition of the synthetic ground water is as follows (in mg/L): Na = 584, K = 7.2, Ca = 328, Mg = 69, SO_4 = 1,626, Cl = 203, NO_3 = 639, and C (inorganic) = 121. The pH was adjusted to about 6.5 and the measured alkalinity was approximately 175 mg/L as $CaCO_3$.

Five percent nitric acid was then used to remove most amorphous oxides that are likely to contain adsorbed contaminants as well as any more resistant water-soluble constituents. As desired, 5 percent nitric acid will not remove contaminants locked in recalcitrant minerals such as apatites or other heavy mineral grains. The acid treatment also dissolves carbonate minerals and releases any sorbed cations. Although oxyhydroxides are stable in most soils, irrigation practices or other land use could cause reducing conditions in the soils and lead to dissolution of the oxyhydroxides with release of their sorbed constituents.

The specific steps in the extraction procedure are as follows:

- Samples were air-dried (no oven heat) and sieved to less than 2 mm.
- Two grams of soil (accurately weighed) was divided between two 50-mL centrifuge tubes; each tube was filled to a 50-mL volume with synthetic ground water solution.
- Tubes were placed in an end-over-end rotary shaker for 4 hours.
- Tubes were removed from the shaker and centrifuged for 30 seconds at 3,000 rpm to remove particles less than 2 microns. Supernatant from both tubes was decanted to a 100-mL volumetric flask and filled to volume with synthetic ground water solution.
- Centrifuge tubes were refilled to 50-mL volume with synthetic ground water solution and placed in an end over end rotary shaker for 30 minutes.
- Tubes were removed from shaker and centrifuged for 30 seconds at 3,000 rpm. Supernatant from both tubes was decanted to a 100-mL volumetric flask and filled to volume with synthetic ground water solution.
- Contents of the two 100-mL volumetric flasks were combined and filtered through a 0.45-micron filter and the filtered sediment residue was retained. Samples were refrigerated for storage prior to submitting for laboratory analysis.

- The same procedure was repeated using the retained sediment samples and a 5 percent nitric acid solution in place of the synthetic ground water.

- All extracted samples were analyzed for As, Mo, Se, U, and V.

From these data, the amount of each constituent removed during each step was calculated.

4.5.3 Subpile Soil Concentration Results

Two grams of soil sample were extracted with a total of 200 mL of lixiviant. The resulting concentrations in micrograms per liter (µg/L) of As, Mo, Se, U, and V measured in each leachate sample were then converted to units of milligrams per kilograms (mg/Kg) in order to estimate the amount of extractable contaminant per mass of subpile soil by the following equation:

$$\frac{200\text{ mL lixiviant}}{2\text{ g soil}} \times \frac{\mu\text{g contaminant}}{\text{L extract}} \times \frac{\text{L}}{1{,}000\text{ mL}} \times \frac{\text{mg}}{1{,}000\,\mu\text{g}} \times \frac{1{,}000\text{ g}}{\text{Kg}} = \text{mg/Kg}$$

For example, a concentration of 1.2 µg/L As was measured in the synthetic ground water extract for soil sample 216–7A. Converting this value to mg/Kg using the above equation results in 0.12 mg of As per Kg of subpile soil that is extractable using synthetic ground water. Using the same equation, concentrations were calculated for the amount of contaminant that is extractable using the nitric acid lixiviant. Contaminant concentrations extractable by the synthetic ground water and nitric acid leach are summarized in Tables 4–7 and 4–8, respectively.

4.6 Water Sampling and Analysis

Each new monitor well was allowed to sit undisturbed for at least 40 hours after final completion before it was developed. Development was performed according to the Drilling Statement of Work. After the wells were properly developed, a total of 53 permanent and temporary monitor wells and 6 surface water locations were sampled from August 10 to August 31, 1998, during a period of average river flow conditions. Additional sampling of 44 permanent monitor wells and five surface water locations was performed from January 20 to January 29, 1999, during low river flow conditions. All samples collected during the 1998 and 1999 sampling events were submitted to the GJO Analytical Laboratory for analyses. The following sections describe the sampling and analysis procedures.

4.6.1 Ground Water Sampling Procedures

Ground water sampling was performed in accordance with the *Sampling and Analysis Plan for the UMTRA Ground Water Project* (DOE 1997b) and the *Environmental Procedures Catalog* (GJO 1998). The following specific procedures from the *Environmental Procedures Catalog* were used for ground water sampling:

- GN–8(P), “Standard Practice for Sample Labeling.”

- GN–9(P), “Standard Practice for Chain-of-Sample-Custody and Physical Security of Samples.”

Table 4–7. Amount of Contaminant Extractable by Synthetic Ground Water per Mass of Subpile Soil

Monitor Well	Sample ID	Sample Depth (ft below surface)	Location	Contaminant (mg/Kg)[a]				
				As	Mo	Se	U	V
Upgradient samples (east of the town of Rifle)								
RFO–292	292–1	8–9.5	Background	0.31	0.12	<0.2	0.38	0.5
RFO–658	658–1	3–9	Background	0.26	0.10	<0.2	0.10	0.5
RFO–659	659–1	23	Background	0.22	<0.10	<0.2	<0.10	<0.1
Mean background				0.26	0.09	0.1	0.18	0.4
Onsite samples								
RFN–216	216–7A	7	Former evaporation pond	0.39	1.1	<0.2	0.76	7.5
RFN–216	216–7B	7	Duplicate	0.42	1.0	<0.2	0.72	7.9
RFN–216	216–11A	11	Former evaporation pond	11.8	15.9	0.86	18.8	293
RFN–216	216–11B	11	Duplicate	12.0	16.7	1.0	19.4	302
Mean 216				6.2	8.7	0.5	9.9	153
RFN–218	218–13A	13	Former tailings area –south	<0.20	1.6	<0.2	0.13	3.1
RFN–218	218–13B	13	Duplicate	0.42	1.8	<0.2	0.13	3.3
Mean 218				0.3	1.7	0.1	0.13	3.2
RFN–225	225–7.5	0–7.5	Former tailings area –north	0.35	1.5	<0.2	0.56	1.1
Mean onsite				2.3	4.0	0.2	3.5	52.4

[a] One-half the detection limit used to calculate the mean

Table 4–8. Amount of Contaminant Extractable by 5 Percent Nitric Acid Solution per Mass of Subpile Soil

Monitor Well	Sample ID	Sample Depth (ft below surface)	Location	Contaminant (mg/Kg)[a]				
				As	Mo	Se	U	V
Upgradient samples (east of the town of Rifle)								
RFO–292	292–1	8–9.5	Background	1.4	0.27	<0.2	0.67	4.3
RFO–658	658–1	3–9	Background	0.99	0.41	<0.2	0.27	1.7
RFO–659	659–1	23	Background	0.82	<0.10	<0.2	0.16	1.5
Mean upgradient				1.1	0.2	0.1	0.2	2.5
Onsite samples								
RFN–216	216–7A	7	Former evaporation pond	2.9	3.4	<0.2	1.5	44.3
RFN–216	216–7B	7	Duplicate	2.5	3.2	<0.2	1.5	42.7
RFN–216	216–11A	11	Former evaporation pond	161	20.7	2.3	64.5	606
RFN–216	216–11B	11	Duplicate	168	20.4	2.2	64.7	614
Mean 216				83.6	11.9	1.2	33.1	327
RFN–218	218–13A	13	Former tailings area –south	1.7	2.7	<0.2	0.62	21.5
RFN–218	218–13B	13	Duplicate	1.5	3.0	<0.2	0.9	24.0
Mean 218				1.6	2.9	0.1	0.8	22.8
RFN–225	225–7.5	0–7.5	Former tailings area –north	2.4	4.4	<0.2	1.1	36.5
Mean onsite				29.2	6.4	0.5	11.7	129

[a] One-half the detection limit used to calculate the mean

- GN–13(P), "Standard Practice for Equipment Decontamination."
- LQ–2(T), "Standard Test Method for the Measurement of Water Levels in Ground Water Monitor Wells."
- LQ–3(P), "Standard Practice for Purging Monitor Wells."
- LQ–4(T), "Standard Test Method for the Field Measurement of pH."
- LQ–5(T), "Standard Test Method for the Field Measurement of Specific Conductance."
- LQ–6(T), "Standard Test Method for the Field Measurement of the Oxidation-Reduction Potential (Eh)."
- LQ–7(T), "Standard Test Method for the Field Measurement of Alkalinity."
- LQ–8(T), "Standard Test Method for the Field Measurement of Temperature."
- LQ–9(T), "Standard Test Method for the Field Measurement of Dissolved Oxygen."
- LQ–10(T), "Standard Test Method for Turbidity in Water."
- LQ–11(P), "Standard Practice for Sampling Liquids."
- LQ–12(P), "Standard Practice for the Collection, Filtration, and Preservation of Liquid Samples."

4.6.2 GJO Analytical Laboratory Sample Analysis

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

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

Analytical methods used for analysis of ground and surface water samples are listed in Table 4–9. Sample preservation consisted of storing the samples in an ice chest with Blue Ice (or equivalent) to cool samples during field sampling, packaging, and shipping. Water samples were analyzed for TDS, major ion constituents, and the following COPCs identified in the draft SOWP: ammonia (as NH_4), arsenic, chloride, fluoride, iron, manganese, molybdenum, nitrate, selenium, sodium, sulfate, uranium, vanadium, lead-210, polonium-210, and thorium-230. Sample handling, preparation, and analyses are described in the references shown in Table 4–9.

Table 4–9. GJO Analytical Laboratory Sample Requirements

Measurement Parameter	Analyte	Sample Container	Analytical Instrument/Method	Detection Limit
Ground and Surface Water	Total uranium	2 each 120 mL	ICP/MS EPA 6020	1.0 µg/L
	Other Inorganics	See Supplemental Water Sampling and Analysis Plan for all UMTRA Sites (DOE 1996d) *and Sampling and Analysis Plan for the UMTRA Ground Water Project* (DOE 1997b).		

Final analytical results were entered into the SEE_UMTRA database and an independent data validation assessment was performed on the 1998 and 1999 sampling events (DOE 1998b and 1999a, respectively). Results of the surface water and ground water analyses are presented in Appendices B and C, respectively.

4.7 Alluvial Aquifer Test

Estimates of the aquifer parameters for the alluvial aquifer are required to develop a better understanding of the hydrogeologic characteristics of the site that could influence migration of contaminants in the ground water and as input parameters in a ground water flow and transport model. Specific details regarding the aquifer test procedures and data analyses are contained in the New Rifle Aquifer Test Calculation Set U0042800. The calculation set also includes copies of the data logger files and curve-matching graphs generated during the analyses of the drawdown and recovery data.

4.7.1 Aquifer Test Procedure

Aquifer tests were completed using newly installed wells RFN–200, –201, and –202 as the pumping wells and surrounding wells as observation wells (Figure 4–1). Two tests were conducted for each well cluster between August and September 1998 and consisted of a pumping and recovery phase, both of which lasted various lengths of time. Observation wells for pumping well RFN–200 were RFN–602, –603, and –195. Observation wells for pumping well RFN–201 were RFN–208 and –196. Observation wells for pumping well RFN–202 were RFN–677, –197, and –675.

The general procedure used for performing the aquifer test consisted of extracting ground water from the pumping wells using an above ground suction lift pump. The discharge line was fitted with a control valve (used to control the pump discharge) immediately followed by a flow meter to monitor the flow rate (in gallons per minute [gpm]) during the pumping period. Pressure transducers were installed in each of the observation wells and used to measure the water level change in each well over the aquifer test periods and throughout the recover tests. Water levels were measured intermittently by hand (using a well sounder) to confirm the accuracy of the pressure transducers. Each pressure transducer was connected to an eight-channel Hermit 3000 data logger that was programmed using In-Situ, Inc., software (Win-Situ, Version 2.13, 1996) from a laptop computer.

The first test performed at each location was a step-test utilizing three or four different pumping rates. The second test for wells RFN–200 and –202 used the optimal pumping rate determined

from the first test, while the second test at well RFN–201 also used three different rates. Specific details regarding pumping rates and test durations are provided in Calculation Set U0042800.

4.7.2 Aquifer Test Analysis

The Theis Method (Theis 1935), the Cooper and Jacob Time-Drawdown method (Cooper and Jacob 1946), and the Neuman Semi-Log Method (Neuman 1975) analyzed all observation well drawdown data. Analysis of the data using Neuman's Semi-Log Method allowed for the estimation of the aquifer specific yield.

If the drawdown data exhibited a delayed yield type of curve, then the data were also analyzed using Neuman's Delayed Yield Method (both the log and semi-log methods are described by Neuman 1975). Drawdown data showing a leaky aquifer type curve were also analyzed using the Hantush Method (Hantush and Jacob 1955). Recovery data were analyzed using the Theis and Jacob Recovery Method (Theis 1935) and the Neuman Semi-Log Recovery Method (Neuman 1975). With the exception of the Neuman Semi-Log Method, all data analyses were completed using curve-matching techniques available through the "AquiferTest" software package (Röhrich and Waterloo Hydrogeologic, Inc., Version 2.52).

4.7.3 Aquifer Test Results

A summary of the aquifer test results is provided in Tables 4–10, 4–11, and 4–12 for data collected at the well clusters RFN–200, –201 and –202, respectively. Data are included for each of the two tests conducted at each well cluster. In most cases results obtained from the first and second tests at a given location are similar. However, for instances in which discrepancies between tests are significant, results of the second test are considered more representative because of its longer duration. Graphic displays of drawdown and recovery data analyses using the various methods described above are included in Calculation Set U0042800.

At the RFN–200 and –201 test locations, a leaky aquifer type of curve resulted from the drawdown data collected from the alluvial observation wells. The data confirm that the source of the leakage was from the underlying aquifer only at the RFN–201 location, insufficient data was available at the RFN–200 test location to determine the source of the leakage. At the RFN–202 test location, the alluvial observation well data resulted in a delayed yield type of curve, suggesting the subsurface conditions may not be consistent across the site.

The ranges of the hydraulic conductivity estimated from the data collected during each aquifer test completed at the wells RFN–200, –201, and –202 locations are summarized in Table 4–13. In addition, the geometric mean is provided along with the associated geometric standard deviation.

Taking into consideration the results generated from the analyses of data collected from tests completed at all three locations the hydraulic conductivity ranges from 53 to 511 feet per day (ft/day) (geometric mean of 123 ft/day). This high end of the range is heavily influenced by the data collected from well RFN–675 (which is located 279 ft from well RFN–202). Excluding the results associated with well RFN–675 the hydraulic conductivity range becomes 53 to 275 ft/day, with a geometric mean of 114 ft/day. While the well 675 data may not provide consistent estimates (compared to the other tests) of the hydraulic parameters, it does indicate that the radius of influence for the well 202 tests exceeded 280 ft.

Table 4–10. Aquifer and Recovery Test Data Analyses Results at RFN–200

Analytical Method	Observation Well 603 Transmissivity (ft^2/day)	Observation Well 603 Hydraulic Conductivity (ft/day)	Observation Well 195 Transmissivity (ft^2/day)	Observation Well 195 Hydraulic Conductivity (ft/day)	Pumping Well 200 Transmissivity (ft^2/day)	Pumping Well 200 Hydraulic Conductivity (ft/day)
TEST 1 (Q = 52.6 gpm)						
Drawdown Data						
Theis	1,800	95	1,138	60	NA	NA
Jacob-Cooper	2,693	142	2,765	145	NA	NA
Hantush	1,014	53	1,276	67	NA	NA
Neuman semi-log	2,184	115	2,320	122	NA	NA
Recovery Data						
Theis & Jacob	2,045	108	1,858	98	1,286	68
Neuman semi-log	2,062	109	1,802	95	1,768	93
TEST 2 (Q = 44.5 gpm)						
Drawdown Data						
Theis	1,211	64	1,915	101	NA	NA
Jacob-Cooper	5,227	275	4,032	212	NA	NA
Hantush	1,211	64	1,512	80	NA	NA
Neuman semi-log	3,652	192	3,739	196	NA	NA
Recovery Data						
Theis & Jacob	1,872	99	2,347	124	1,094	58
Neuman semi-log	1,963	103	2,574	136	1,083	57

Notes: Q = Discharge
NA = data not analyzed to determine hydraulic parameter

Table 4–11. Aquifer and Recovery Test Data Analyses Results at RFN–201

Analytical Method	TEST 1 (Q = 15.7 gpm)				TEST 2 (Q = 12.5 gpm)			
	Observation Well 196 Transmissivity (ft^2/day)	Observation Well 196 Hydraulic Conductivity (ft/day)	Pumping Well 201 Transmissivity (ft^2/day)	Pumping Well 201 Hydraulic Conductivity (ft/day)	Observation Well 196 Transmissivity (ft^2/day)	Observation Well 196 Hydraulic Conductivity (ft/day)	Pumping Well 201 Transmissivity (ft^2/day)	Pumping Well 201 Hydraulic Conductivity (ft/day)
Drawdown Data								
Theis	1,699	169	NA	NA	2,693	266	NA	NA
Jacob-Cooper	1,800	177	NA	NA	2,362	233	NA	NA
Hantush	1,699	169	NA	NA	1,901	189	NA	NA
Neuman semi-log	2,409	239	NA	NA	2,262	224	NA	NA
Recovery Data								
Theis & Jacob	1,656	164	812	80	1,656	164	924	92
Neuman semi-log	1,606	159	852	84	1,575	156	630	62

Notes: Q = Discharge
NA = data not analyzed to determine hydraulic parameter

Table 4–12. Aquifer and Recovery Test Data Analyses Results at RFN–202

Analytical Method	TEST 1 (Q = 93.4 gpm)				TEST 2 (Q = 80.5 gpm)					
	Observation Well 197 Transmissivity (ft^2/day)	Observation Well 197 Hydraulic Conductivity (ft/day)	Pumping Well 202 Transmissivity (ft^2/day)	Pumping Well 202 Hydraulic Conductivity (ft/day)	Observation Well 197 Transmissivity (ft^2/day)	Observation Well 197 Hydraulic Conductivity (ft/day)	Observation Well 675 Transmissivity (ft^2/day)	Observation Well 675 Hydraulic Conductivity (ft/day)	Pumping Well 202 Transmissivity (ft^2/day)	Pumping Well 202 Hydraulic Conductivity (ft/day)
Drawdown Data										
Theis	3,586	246	NA	NA	3,096	213	7,416	511	NA	NA
Jacob-Cooper	1,829	126	NA	NA	1,714	119	3,470	239	NA	NA
Neuman log	1,800	124	NA	NA	1,728	120	7,416	511	NA	NA
Neuman semi-log	1,533	106	NA	NA	1,457	101	4,370	301	NA	NA
Recovery Data										
Theis & Jacob	1,384	96	1,319	91	1,146	79	2,074	143	1,164	80
Neuman semi-log	1,318	91	1,221	84	1,184	82	2,066	142	1,072	74

Notes: Q = Discharge
NA = data not analyzed to determine hydraulic parameter

Table 4–13. Statistical Results From The New Rifle Aquifer Tests (all values in ft/day)

	Well RFN–200			Well RFN–201			Well RFN–202				
	Test 1	Test 2	Test 1 & 2	Test 1	Test 2	Test 1 & 2	Test 1	Test 2	Test 2[a]	Test 1 & 2	Test 1 & 2[a]
Hydraulic Conductivity Range	53–145	57–275	53–275	80–239	62–266	62–266	68–246	74–511	74–213	68–511	68–246
Geometric Mean	94	111	102	146	158	152	108	155	102	135	105
Standard Deviation	29	68	53	52	70	60	56	150	46	128	50

[a]Excludes results at observation well RFN–675

4.8 Surface Flow Measurements

The alluvial aquifer underlying the New Rifle site is partially recharged by surface water from Pioneer ditch and from four unnamed tributaries entering the site from the north. The four tributaries flow south and eventually discharge into the Colorado River.

Flow measurements were performed January 1999 to determine the volume of surface water entering the alluvial aquifer from Pioneer ditch and these tributaries. Measurements were taken at three locations along Pioneer ditch. Surface water flow rates of approximately 94.2 gpm, 65.9 gpm, and 25.1 gpm were observed at these locations. The distance between the first and second locations is approximately 2,200 ft and the distance between the second and third locations is approximately 3,100 ft. The differences of 28.3 gpm and 40.8 gpm represents the amount lost to infiltration into the alluvial aquifer from the unlined ditch between the respective locations. Dividing the loss by the distance between the locations results in a loss of approximately 0.013 gpm for each foot. The length of Pioneer ditch starting at the east end of the site to the point of discharge into the Colorado River west of the Roaring Fork pond is approximately 7,100 ft, which results in a net recharge of 92.6 gpm (17,821 cubic feet per day

[ft^3/day]) to the aquifer. A single measurement just above the confluence of Pioneer ditch and the first tributary west of the site indicates that the tributary flow is approximately 8.4 gpm.

4.9 Ecological Investigation

Characterization of the ecology of the New Rifle site was needed to complete the assessment of ecological risks associated with site-related contaminated ground water. A defensible ecological risk assessment will support the development of a risk-based compliance strategy. In general, the objective of the field investigation was to collect data adequate to evaluate potential exposure pathways and receptors at the New Rifle site. Activities required included:

- Characterization of current and possible future plant ecology. Of particular importance was the identification of any phreatophyte species rooted into areas of the ground water plume with highest contaminant concentrations.

- Sampling and chemical analysis of phreatophyte plant tissue in contaminated areas and comparison of these results with those for reference areas (i.e., areas outside the influence of site activities). Results of chemical analysis are to be used to calculate hazard indices (HIs) and HI ratios for toxicity to the plants and to the animals that might ingest them.

- Sampling and chemical analysis of COPCs in the mitigation wetland, Roaring Fork holding pond, and reference area sediment and surface water. Results of chemical analysis are to be used to calculate HIs and HI ratios for aquatic life and other representative receptors.

The following sections briefly describe the ecological field activities and methodology.

4.9.1 Plant Ecology Characterization

Plants that root into the plume or are irrigated with plume water are potential exposure pathways for humans and ecological receptors. The vegetation also influences recharge and discharge components of the hydrologic system. Current and possible future plant ecology at the former New Rifle millsite and tailings area were investigated as part of evaluations of (1) ecological risks associated with site-related contaminated ground water, and (2) the relative importance of on-site evapotranspiration as a component of the site water balance.

4.9.1.1 Methods

The semi-quantitative relevé technique (Bonham 1989) was used to characterize species composition and relative abundance of plant communities of the former millsite and tailings area by subjectively selecting representative stands of each vegetation type, walking through the stands, and compiling a list of all species encountered. Each species was then assigned one of six cover classes. Cover was not measured precisely. The site was traversed on June 2, 1998. Results of the traverse are presented below.

4.9.1.2 Field Traverse

Results of the June 2, 1998, field traverse indicate that the New Rifle mitigation wetland contains phreatophytic plants (plants that root in ground water). All of the willows (*Salix sp.*), cottonwood (*Populus fremontii*), cattails (*Typha latifolia*), and tamarisk (*Tamarix ramosissima*) are

phreatophytes. These plants create potential exposure pathways at the New Rifle site. The field traverse of the millsite area revealed seven different vegetation types (Table 4–14). Three are upland plant communities. The other four are wetland plant communities.

Table 4–14. Vegetation Types at New Rifle Site

Taxonomic Name	Common Name	Tall Wheat Grass	Cattail/ Shrub	Foxtail/ Alkali Grass	Foxtail Meadow	Kochia	Alkali Grass	Upland
Agropyron elongatum	Tall wheatgrass	4		+		1		3
Alopecuris arundinaceus	Creeping foxtail			3	5	2	1	2
Artemesia tridentata	Sagebrush							+
Bromus inermis	Smooth brome	+		+				+
Bromus tectorum	Cheatgrass	2		+		1		
Centaurea repens	Russian knapweed							+
Convulvulus arvensis	Field bindweed							+
Deschampsia cespitosa	Tufted hairgrass		+					
Descurania pinnata	Tansy mustard	1				+	+	1
Hordeum jubatum	Foxtail barley		1					1
Kochia scoparia	Kochia	2		1	1	3	1	2
Lepidium montanum	Western pepperweed	1						
Lepidium perfoliatum	Clasping pepperweed			+		+		
Melilotus officionale	Yellow sweet clover			+				+
Muhlenbergia asperifolia	Alkali muhly		1		+			
Oenothera sp.	Evening primrose							+
Opuntia polycantha	Prickly pear cactus							+
Oryzopsis hymenoides	Indian ricegrass							1
Populus fremontii	Fremont's cottonwood		1	+				
Puccinellia distans	Fults alkaligrass		+	3	+	2	5	
Salix exigua	Sandbar willow		1					
Salix lasiandra	Whiplash willow		+					
Salix planifolia	Planeleaf willow		+					
Salsola kali	Russian thistle	+						1
Scirpus americanus	Olneys three-square		1					
Sporobolis airoides	Alkali sacaton		+					
Tamarix ramosissima	Salt cedar/tamarisk		4	1			1	
Tragapogon dubius	Western salsify	+				+		+
Trifolium sp.	Clover			+				
Typha latifolia	Cattail		2					
Ulmus pumila	Chinese elm		+					

COVER CLASSES: (+) <1%, (1) 1-5%, (2) 5-25%, (3) 25-50%, (4) 50-75%,(5) 75-100%

The following is a brief description of each of the vegetation types at the New Rifle site:

- The tall wheatgrass type covers most of the New Rifle site. It is composed of mostly tall wheatgrass, kochia, and cheatgrass.

- The cattail/shrub wetland is found on the east end of the wetland. The dominant plant species are tamarisk, cattails, and willows. This is the wettest area of the mitigation wetland. The diversity of this area is higher than anywhere else on the site.

- The foxtail/alkaligrass type covers a large proportion of the mitigation wetland. These are the two wetland species that were seeded and they occur together in this area.

- The foxtail meadow is a small wet area that is totally dominated by creeping foxtail.
- The kochia type is a weedy area on the northwest end of the mitigation wetland, which is too dry for wetland plants to thrive. The resulting vegetation is dominated by kochia and other weeds.
- The alkaligrass type is an area on the west end of the wetland that was seeded at a later date than the rest of the wetland. It is dominated by Fults alkaligrass.

In the absence of disturbance, future upland plant communities at the site should trend toward a shrubland dominated by either greasewood (*Sarcobatus vermiculatus*) or rabbitbrush (*Chrysothamnus viscidiflorus*). The future vegetation of the east half of the wetland is a cattail/shrub wetland, including tamarisk, and willows. There is also a wet meadow area on the east end of the wetland dominated by creeping foxtail and Fults alkaligrass. The western half of the wetland mitigation on the site was reconstructed in May 1999. The future vegetation of the west end of the wetland is designed to eventually become a combination of shrub wetland, wet meadow, and emergent wetland dominated by bulrushes.

4.9.2 Sampling for Chemical Analysis

Field sampling to support the BLRA update was conducted in May 1998. Sample collection for chemical analysis included surface water and sediment at the New Rifle mitigation wetland, Roaring Fork holding pond, and a reference wetland. Vegetation sampling for chemical analysis was limited to the New Rifle mitigation wetland and reference wetland. These activities were identified in the work plan (DOE 1998c). Reconnaissance visits to the New Rifle site and the Roaring Fork holding pond indicated that ecological sampling was best focused at the New Rifle mitigation wetland area and the larger, less disturbed westernmost holding pond.

4.9.2.1 Selection of a Reference Location

The selection of an appropriate location for a reference area with wetland characteristics was necessarily limited to two areas upgradient of the former Old Rifle site. The gravel pit ponds just south of One Mile Pond had small areas with some wetlands characteristics but were being actively mined. In addition, the availability of desired phreatophytic species was limited. Sediment sampling would have been constrained by the presence of large amounts of rock and cobble. The alternate location (One Mile Pond) proved to be more satisfactory for ecological sampling due to the natural wetland conditions present, observed wildlife species, wetlands vegetation, accessibility, and presumed continued and undisturbed existence (Figure 4–6). Analytical data were also available for One Mile Pond that has historically been considered a background location for the Rifle site.

4.9.2.2 Abiotic Sampling

Surface water and sediment samples were collected from May 26 through May 29, 1998, at the New Rifle wetland, an upgradient reference wetland (One Mile Pond), and the westernmost Roaring Fork pond (Figure 4–6). Five co-located samples of sediment and surface water were collected at each of five locations at each site. The number of samples was chosen to satisfy a coefficient of variation of 20, a minimum detectable relative difference (MDRD) of 20, a

confidence of 80 (Type I error, false positive), and a power of 90 (Type II error, false negative) based on a 1-sided, single sample distribution. This is consistent with EPA guidance (EPA 1989b). Other factors which were considered in the selection of sample size were the small areal extent of the affected sites, the amount of historical data available, generally low contaminant levels, and budgetary constraints. The surface water samples were collected as grab samples while the sediment samples were assumed to consist of materials collected from a nominal depth of 0 to 6 inches below the sediment surface. Surface water sample collection preceded sediment and biota tissue sample collection. All surface water and sediment sampling containers were obtained precleaned from an industrial supplier and accompanied by a cleanliness certificate.

Sample locations were identified as follows:

- Locations 1201–1205—Reference wetland (One Mile Pond)
- Locations 1206–1210—New Rifle mitigation wetland
- Locations 1211–1215—Roaring Fork holding pond

Sample locations at the Roaring Fork holding pond were restricted to areas along the southeastern shore and southern boundary of the westernmost pond due to safety concerns associated with mining operations and bank stability. Field sample locations at the New Rifle wetland and Roaring Fork holding pond were surveyed and converted into state plane coordinates, while the reference wetland locations were established by tape and compass.

Surface Water Samples

Both filtered and unfiltered surface water samples were collected at the same locations as the sediment samples. The filtered component represents the soluble component for aquatic receptors, while terrestrial receptors ingest unfiltered surface water. Filtered surface water samples were identified with an “F” suffix to the sample identification number while unfiltered samples received a “U” (unfiltered) suffix. Each sampling bottle was first rinsed with the surface water and that portion discarded prior to sample collection. The sample was collected by immersing the bottle just below the water surface and filling to just below the lid. Samples were then filtered using a 0.45 µm filter and acidified accordingly. Table 4–15 provides a summary of analytes, preservatives, containers, and other information pertaining to surface water sample collection.

Sample labels showing the date, time, location, laboratory bar code, sampler, analyses requested, preservatives, and comments were applied to each container and secured with clear plastic tape. All sample containers were placed in coolers containing ice until transported to the GJO Analytical Chemistry Laboratory. A chain of custody (CoC) form was completed for all samples and a CoC label placed over each cooler. All samples were maintained under strict CoC.

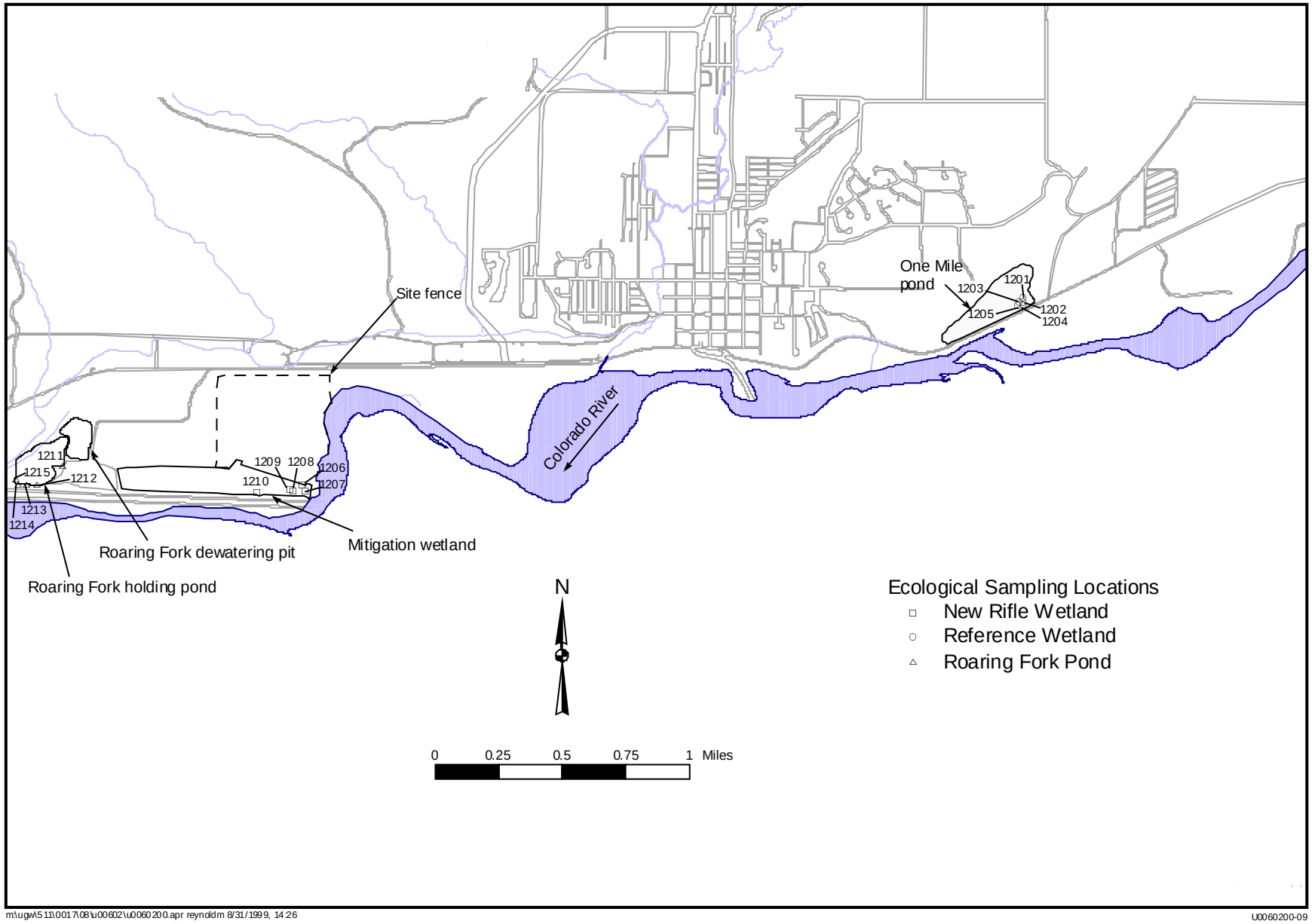


Figure 4–6. Ecological Sample Locations

Table 4–15. Summary of Surface Water Sampling Parameters

Analyte	Preservative	Container	Holding Time	Method
Arsenic	HNO_3 – pH <2; cool 4 °C	500 mL amber HDPE	6 months	ICPAES
Cadmium	HNO_3 – pH <2; cool 4 °C	500 mL amber HDPE	6 months	ICPMS
Iron	HNO_3 – pH <2; cool 4 °C	500 mL amber HDPE	6 months	ICPAES
Manganese	HNO_3 – pH <2; cool 4 °C	500 mL amber HDPE	6 months	ICPAES
Molybdenum	HNO_3 – pH <2; cool 4 °C	500 mL amber HDPE	6 months	ICPMS
Selenium	HNO_3 – pH <2; cool 4 °C	500 mL amber HDPE	6 months	ICPAES
Strontium	HNO_3 – pH <2; cool 4 °C	500 mL amber HDPE	6 months	ICPAES
Uranium	HNO_3 – pH <2; cool 4 °C	500 mL amber HDPE	6 months	ICPMS
Vanadium	HNO_3 – pH <2; cool 4 °C	500 mL amber HDPE	6 months	ICPAES
Zinc	HNO_3 – pH <2; cool 4 °C	500 mL amber HDPE	6 months	ICPAES
Ammonia as Ammonium (NH_4)	H_2SO_4 – pH <2; cool 4 °C	125 mL HDPE	28 days	SPEC
Nitrate	H_2SO_4 – pH <2; cool 4 °C	125 mL HDPE	28 days	IC
Phosphate	H_2SO_4 – pH <2; cool 4 °C	125 mL HDPE	28 days	SPEC
Fluoride, Sulfate	cool 4 °C only	125 mL HDPE	28 days	IC

HDPE– high density polyethylene
H_2SO_4–sulfuric acid
HNO_3–nitric acid
mL– milliliter
C–centigrade or Celsius
ICPMS–Inductively-coupled plasma-mass spectrometry
ICPAES–Inductively-coupled plasma-atomic emission spectroscopy
IC–Ion chromatography
SPEC–Spectroscopy

Sediment Samples

Each sediment sample represented a composite of typically three or four locations where vegetation material was present. The area for collection was typically a circle of radius less than 5 ft. Excess organic matter and larger rocks and pebbles were removed from the sample prior to compositing. The contents of one stainless-steel auger (i.e., one subsample) was collected at each composite location and placed in a large stainless steel mixing pan. All subsamples were mixed thoroughly with a stainless steel spoon prior to removing approximately 4 ounces (114 grams) of material for metals analysis. In addition, a 125-mL HDPE bottle was collected for ammonia, nitrate, and phosphate, and another for fluoride and sulfate. Table 4–16 provides a summary of analytes, preservatives, containers, and other information pertaining to sediment sample collection.

Table 4–16. Summary of Sediment Sampling Parameters

Analyte	Preservative	Container	Holding Time	Method
Arsenic	cool 4 °C	4 ounce amber glass	6 months	ICPAES
Cadmium	cool 4 °C	4 ounce amber glass	6 months	ICPMS
Iron	cool 4 °C	4 ounce amber glass	6 months	ICPAES
Manganese	cool 4 °C	4 ounce amber glass	6 months	ICPAES
Molybdenum	cool 4 °C	4 ounce amber glass	6 months	ICPMS
Selenium	cool 4 °C	4 ounce amber glass	6 months	ICPAES
Strontium	cool 4 °C	4 ounce amber glass	6 months	ICPAES
Uranium	cool 4 °C	4 ounce amber glass	6 months	ICPMS
Vanadium	cool 4 °C	4 ounce amber glass	6 months	ICPAES
Zinc	cool 4 °C	4 ounce amber glass	6 months	ICPAES
Ammonia as NH_4	cool 4 °C	125 mL HDPE	28 days	SPEC
Nitrate	cool 4 °C	125 mL HDPE	28 days	IC
Phosphate	cool 4 °C	125 mL HDPE	28 days	SPEC
Fluoride, Sulfate	cool 4 °C	125 mL HDPE	28 days	IC

Sample labels showing the date, time, location, laboratory bar code, sampler, analyses requested, preservatives, and comments were applied to each container and secured with clear plastic tape. All sample containers were placed in coolers containing ice until transported to the GJO Analytical Chemistry Laboratory. A CoC form was completed for all samples and a CoC label placed over each cooler. All samples were maintained under strict CoC.

4.9.2.3 Quality Assurance Samples

Field blanks and equipment rinses were collected at both the reference wetland and Roaring Fork holding pond. These samples consisted of distilled, deionized water appropriately preserved and cooled in the field. The field blank was prepared by pouring distilled, deionized water directly from the carboy into the appropriate sampling bottle and preserving as necessary. The equipment rinse consisted of pouring distilled deionized water from the carboy over the cleaned sampling equipment (auger, sampling pan, shears, and spoons) and collecting the rinsate in the appropriate sampling containers and preserving and cooling as necessary. Due to the close proximity between the Roaring Fork holding pond and the New Rifle mitigation wetland, and the small number of samples, no additional equipment rinse and field blank were collected at the New Rifle site.

A field duplicate surface water sample was collected at reference wetland location No. 1203. In addition, field duplicate surface water samples were collected at New Rifle wetland location No. 1208 and Roaring Fork holding pond location No. 1215. Field duplicates were identified with “D” (duplicate) suffix appended to the sample identification number.

A field duplicate sediment sample was collected at reference wetland location No. 1203. In addition, field duplicate sediment samples were collected at New Rifle wetland location No. 1210 and Roaring Fork holding pond location No. 1215. Field duplicates were identified with an “D” (duplicate) suffix appended to the sample identification number.

4.9.2.4 Biotic Sampling

Vegetation samples consisting of cattails (*Typha* sp.) were collected at both the reference and New Rifle wetland locations. The need to collect biota tissue samples for chemical analysis was identified in the screening-level risk assessment (DOE 1996a), SOWP (DOE 1996c), and work plan (DOE 1998c). These samples were co-located with the surface water and sediment samples. No vegetation was collected at the Roaring Fork holding pond in accordance with the work plan. Samples were collected by digging up an entire plant or cluster of plants using a stainless steel shovel. Excess sediment was rinsed off the plants prior to separating the roots and stems. Stems and roots were processed with pruning shears having stainless steel and polyethylene cutting edges. The roots and stems were rinsed thoroughly with sample water, followed by tap and distilled deionized water rinses until rinsates contained no visible soil or sand particles. All plant materials received a final distilled, deionized water rinse prior to bagging. Stems and roots were composited separately from three or four samples depending on the size of the original plant or plant cluster. Stems and roots were doubly bagged in clean zip-loc type storage bags. Sample labels were applied to each the outermost zip-loc type bag and secured with clear plastic tape. All samples were kept in coolers containing ice for transportation to the GJO Analytical Chemistry Laboratory. A CoC form was completed for all samples and a CoC label placed over each cooler. All samples were maintained under strict CoC. Samples which could not be processed directly at the laboratory by freeze-drying were placed in freezers at 4 °C. Table 4–17

provides a summary of analytes, preservatives, containers, and other information pertaining to biota tissue collection.

Table 4–17. Summary of Cattail Sampling Parameters

Analyte	Preservative	Container	Holding Time	Method
Arsenic	Cool 4 °C	double 1-gal zip-loc type bags	6 months	ICPAES
Cadmium	Cool 4 °C	double 1-gal zip-loc type bags	6 months	ICPMS
Iron	Cool 4 °C	double 1-gal zip-loc type bags	6 months	ICPAES
Manganese	Cool 4 °C	double 1-gal zip-loc type bags	6 months	ICPAES
Molybdenum	Cool 4 °C	double 1-gal zip-loc type bags	6 months	ICPMS
Selenium	Cool 4 °C	double 1-gal zip-loc type bags	6 months	ICPAES
Strontium	Cool 4 °C	double 1-gal zip-loc type bags	6 months	ICPAES
Uranium	Cool 4 °C	double 1-gal zip-loc type bags	6 months	ICPMS
Vanadium	Cool 4 °C	double 1-gal zip-loc type bags	6 months	ICPAES
Zinc	Cool 4 °C	double 1-gal zip-loc type bags	6 months	ICPAES

gal = gallon(s)

Sample locations were identified as follows:

- Locations 1201–1205—Reference wetland
- Locations 1206–1210—New Rifle mitigation wetland

Vegetation samples were uniquely identified by adding an “R” (root) or “S” (stem) suffix to each sample identification. All sample bags for each field location containing the same laboratory identification number were processed as one sample (i.e., all roots for the same sample identification and field location number were processed as one sample). The same procedure applied to the stem material for each field sampling location.

Quality Assurance Samples

A field duplicate cattail sample was collected at reference wetland location No. 1203. In addition, another field duplicate cattail sample was collected at New Rifle wetland location No. 1210. Field duplicates were identified with a “D” (duplicate) suffix appended to the sample identification number. Equipment rinsates and field blanks identified in Section 4.9.2.3 applied to the biota collection as well.

4.9.3 Other Potential Ecological Receptors

The work plan for the Rifle sites (DOE 1998c) identified potential terrestrial and aquatic receptors that could be exposed to contamination at the New Rifle site. This information was derived from qualitative surveys, observations, and previous reports. More specific data was collected regarding site-specific vegetation, as described above. No additional surveys were conducted to identify site-specific wildlife and aquatic organisms; existing data were determined to be sufficient in for development of the conceptual model and ecological risk assessment. These data are summarized in this section.

Lists of potential wildlife receptors at the New Rifle site (Tables 4–18 and 4–19) are based on observations before remediation and on species expected to be attracted to the new habitats

created. A few of these species have been observed at the site. Most were included after a review of the applicable literature (Hammerson 1986; Van Velzen 1980).

No thorough surveys of aquatic organisms in surface-water bodies near the New Rifle site has been conducted. However, the following organisms were observed in the wetland ditch at the New Rifle site before the remedial action: water striders, backswimmers, mosquito larvae, and midge larvae (DOE 1996a). Game fish known to inhabit the area include green sunfish, black bullhead, brown trout, and rainbow trout. Bluehead sucker, flannelmouth sucker, common carp, roundtail chub, and fathead minnow also occur in the area.

Table 4–18. Mammals, Amphibians, and Reptiles Expected to Inhabit the Mitigation Wetlands at the New Rifle Site

Mammals	Amphibians and Reptiles
Muskrat	Woodhouse toad
Raccoon	Northern leopard frog
Mule deer	Racer
Rabbits	Corn snake
Hares	Bullsnake
Vole species	Western terrestrial garter snake
Mice species	

Table 4–19. Breeding Birds That May Nest in the Mitigation Wetlands at the New Rifle Site

Pied-Billed Greb	Killdeer	American Robin
Great Blue Heron	Common Snipe	European Starling
Black-Crowned Night Heron	Spotted Sandpiper	Yellow Warbler
Canada Goose	Common Nighthawk	Common Yellowthroat
Mallard	Belted Kingfisher	Yellow-Breasted Chat
Gadwall	Western Kingbird	Green-Tailed Towhee
Pintail	Say's Phoebe	Rufus-Sided Towhee
Green-Winged Teal	Willow Flycatcher	Savannah Sparrow
Blue-Winged Teal	Olive-Sided Flycatcher	Chipping Sparrow
Cinnamon Teal	Barn Swallow	Song Sparrow
American Widgeon	Cliff Swallow	Yellow-Headed Blackbird
Common Merganser	Black-Billed Magpie	Red-Winged Blackbird
Northern Harrier	Common Crow	Northern Oriole
American Kestrel	Dipper	Brewer's Blackbird
Virginia Rail	Bewick's Wren	Black-Headed Grosbeak
Sora	Northern Mockingbird	American Goldfinch
American Koot	Gray Catbird	Lesser Goldfinch

5.0 Site Conceptual Model

This section presents an interpretation of the site characterization data collected in 1998, integrates the most recent data with previous information, and provides the most current understanding of the extent and magnitude of contamination and exposure pathways to the public health and environment. The conceptual model provides the basis for the subsequent risk assessment (Section 6.0) and the proposed ground water compliance strategy (Section 7.0).

5.1 Geology

Principal geologic rock types in the site area include a diverse assemblage of unconsolidated Quaternary deposits and Tertiary age sedimentary beds of the Wasatch Formation. The unconsolidated deposits consist mostly of alluvial silt, sand, and cobble gravels in stream channels and beneath flood plains and in terraces along the Colorado River Valley and its major tributaries, fine-grained silt and sand overbank deposits, and mass wasting deposits consisting of sheetwash, colluvial, and alluvial fan material. The Wasatch Formation consists mostly of variegated claystone, siltstone, and fine-grained sandstone of fluvial origin. The former New Rifle milling site is situated on a broad section of Colorado River floodplain alluvium deposited over several thousand feet of Wasatch Formation.

Ground water in the alluvial aquifer and the shallow Wasatch Formation directly beneath the site are the hydrostratigraphic units of primary interest. Therefore, the geologic field investigations concentrated on characterizing the nature and extent of the alluvial material and the relationship between the alluvium and the shallow Wasatch rocks that are most susceptible to vertical contaminant invasion from the site. Presented in this section is a brief overview of the regional geologic setting followed by site-specific geologic information. These data form the basis for the site conceptual hydrostratigraphic model that supports the ground water flow and transport computer models used to simulate natural flushing.

5.1.1 Regional Geologic Setting

Structurally, the site is located near the southeastern edge of the Piceance Creek basin and along the southwestern edge of the Grand Hogback monocline that was formed in response to the adjacent White River uplift. Numerous faults expose Cambrian- to Tertiary-age bedrock on the steeply dipping rock surfaces as shown in the regional geologic map presented in Figure 5–1. A north to south geologic cross-section of the Rifle region is presented in Figure 5–2. Brief overviews of the principal rock units, from oldest to youngest, that are present in the region are described below.

The oldest rock units in the region are present north of Rifle, between the Grand Hogback and the White River uplift, where a 6-mile section of near-vertical sedimentary beds of Paleozoic and Mesozoic age are exposed. Located in this area is one of the largest vanadium-uranium deposits on the Colorado Plateau that occurs in host rocks of the Triassic Chinle Formation, Triassic-Jurassic Glen Canyon Sandstone, and Jurassic Entrada Sandstone (Chenoweth 1982, Fischer 1960). The deposits produced approximately 47 million pounds of V_2O_5 and about 1 million pounds of U_3O_8 from the Garfield and Rifle Mines from 1925 through 1977. A few miles east of the Rifle mine several smaller vanadium-uranium deposits were mined from the Salt Wash Member of the Morrison Formation.

Sediments containing uranium, vanadium, and other metals from these mineralized Triassic-Jurassic host rocks were eroded and transported south and deposited in the Wasatch Formation prior to the later part of the Laramide deformation when intensive folding of the Grand Hogback monocline occurred in response to the White River uplift. As a result of the uplift the sedimentary beds of the Wasatch Formation are now at a near-vertical orientation along the face of the Grand Hogback monocline. To the southwest of the monocline the dip declines rapidly to 30 to 40 degrees just north of Rifle and then flattens to a gentle dip of 5 degrees or less in the vicinity of the site. Shallow bedrock between the Colorado River and the Grand Hogback consists of variegated shale, siltstone, and fine-grained sandstone of the Wasatch Formation. Although the Wasatch Formation contains some resistant beds that form cliffs, most of the formation weathers easily and has formed lowland mesas, including the Webster and Prefontaine Mesas directly west to northwest of the city of Rifle.

Younger sedimentary rocks overlying the Wasatch Formation are the Tertiary sandstone, oil shale, and marlstone beds of the Green River Formation which crop out almost continuously around the southeastern margin of the Piceance Creek Basin forming the prominent Roan Cliffs. These cliffs are visible to the northwest from the town of Rifle.

The youngest geologic units include a diverse assemblage of Quaternary alluvium, colluvium, landslides, debris-flow, and loess that has been mapped in and adjacent to the Colorado River valley near Rifle (Shroba et al. 1995, Stover 1993). These surficial unconsolidated units overlie several thousand feet of Tertiary-age Wasatch Formation.

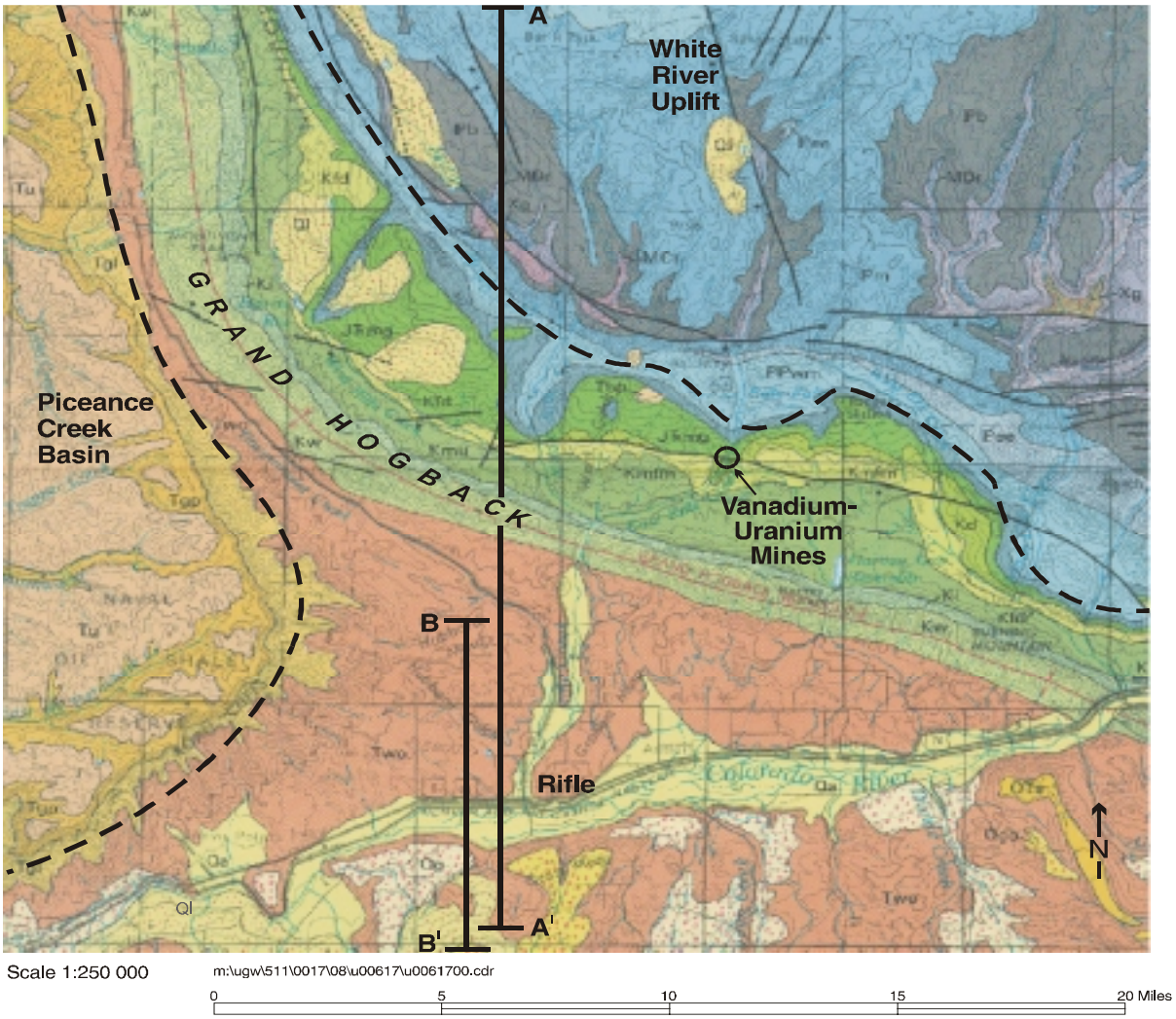
5.1.2 Local Geology and Stratigraphy

The New Rifle site is located on a broad section of Colorado River floodplain alluvium deposited over several thousand feet of Wasatch Formation. A generalized map showing the surficial geology in the area of the site is presented in Figure 5–3. Observable characteristics of the sedimentary rocks of the Wasatch Formation and the alluvial deposits that are exposed or penetrated by boreholes at the site are described below. These are the primary rock types affecting the ground water at the site.

5.1.2.1 Tertiary Wasatch Formation

Sediments in the Wasatch Formation were deposited in a braided stream, floodplain lacustrine, and a lacustrine environment during the Paleocene and early Eocene. Donnell (1969) describes the Formation as a single stratigraphic unit that is divided into three members. The stratigraphic relationship of the three members beneath the site are illustrated in the geologic cross-section presented in Figure 5–4. The upper and lower members, the Shire and Atwell Gulch, respectively, consist mostly of variegated claystones. The intervening Molina Member consists predominantly of a thick brown continuous ledge-forming massive arkosic sandstone with only minor amounts of thin interbeds of variegated claystone.

The stratigraphic equivalents of the three members crop out in entirety along the Grand Hogback approximately 6 miles north of the town of Rifle. The stratigraphic equivalent of the Shire Member exposed along the Grand Hogback is in excess of 4,000 ft thick. The thickness of the Shire Member is highly variable with an overall thinning to approximately 1,800 ft toward the west (Donnell 1969).



Qa	Unconsolidated deposits (Holocene): Alluvium	Tgp	Parachute Creek Member (Eocene)	Kd	Dakota Sandstone (Lower Cretaceous)
Ql	Landslide deposits (Holocene and Pleistocene) large slump blocks of basalt irregularly veneered with glacial drift	Tgl	Lower part of Green River Formation (Eocene)	Kfd	Frontier Sandstone and Mowry Shale Members of Mancos Shale and Dakota Sandstone (Lower Cretaceous)
Qg	Unconsolidated deposits (Pleistocene) young gravels (Bull Lake and younger)	Two	Wasatch and Ohio Creek Formations (Eocene and Paleocene)	JTRmg	Morrison, Entrada, and Glen Canyon Formations (Upper Jurassic to Upper Triassic)
Qta	High-Level Alluvium (Pleistocene and/or Pliocene)	Kw	Williams Fork Formation (Upper Cretaceous)	PIPwm	Weber Sandstone and Maroon Formation
Tbp	Browns Park Formation (Miocene)	Ki	Iles Formation (Upper Cretaceous)	IPm	Minturn Formation (Pennsylvanian)
Tu	Uinta Formation (Eocene)	Kmu	Upper unit of Mancos Shale (Upper Cretaceous)	IPb	Belden Formation (Pennsylvanian)
		Kmfm	Frontier Sandstone and Mowry Shale Members and intervening shale zone (Upper and Lower Cretaceous)	M€r	Mississippian, Devonian, Ordovician, and Cambrian Rocks

Figure 5–1. Regional Geologic Map Showing Principal Physiographic Features
(modified from Tweto et al. 1978)

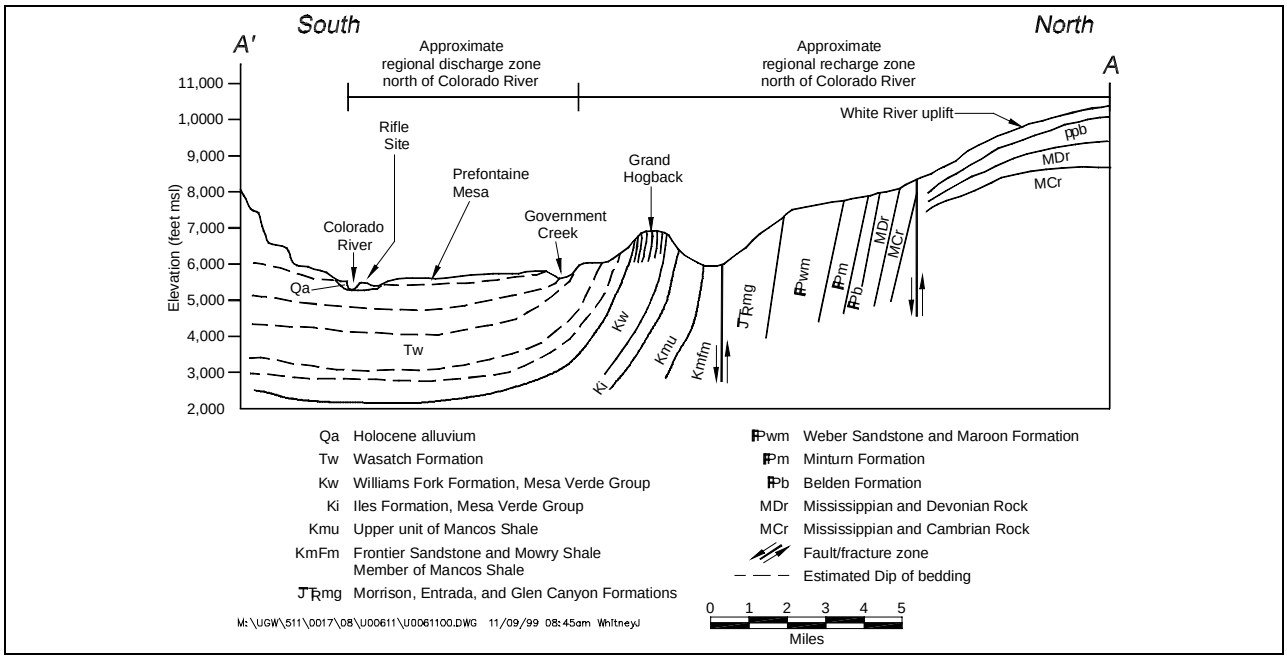


Figure 5–2. Regional Geologic Cross-Section of the Rifle Area

Wasatch sediments directly beneath the site are probably correlative to the lower portion of the Shire Member as most of the member has been removed by erosion. The general bedrock topography of the erosional surface in the vicinity of the site is illustrated in Figure 5–5 by the contour lines drawn at 50-ft intervals on top of the Wasatch Formation. The bedrock contour map was prepared from borehole information, USGS topographic maps, and field mapping conducted during the site investigation (Section 4.0). In general, the erosional surface of the Wasatch Formation beneath the site reflects a fairly flat topographic surface that gently slopes to the west-southwest. Contour elevations abruptly steepen to the north and south reflecting areas where the Shire Member crops out above the Colorado River floodplain. Rock core samples cut from the Wasatch Formation beneath the site indicate the Shire Member consists of variegated colored claystones and siltstones with minor units of brown lenticular sandstones. The thickness of the Shire Member beneath the site is unknown, but it is at least 149-ft thick as evidenced in the deepest boring at the site (RFN–640).

Donnell (1969) mapped a possible equivalent bed of the middle Molina Member that is approximately 400 ft thick at Rifle Gap. Another possible stratigraphic equivalent unit of the middle Molina Member attains a thickness of 600 ft in the Carter 1 E.C. Hunter gas well (T6S, R94W) approximately 5 miles southwest of Rifle. The sandstone beds penetrated by this well are thin and separated by thick units of siltstone and variegated claystone. The Molina Member has not been encountered in any boring at or near the site, but is probably about 500 ft thick.

At Rifle Gap a possible stratigraphic equivalent unit of the lower Atwell Gulch Member is approximately 700 ft thick (Donnell 1969). The lower member is probably about 600 ft thick in the area of the site. Locally, the lower member consists of a series of brown and gray shales and sandstones with several thin, discontinuous interbeds of lignite and carbonaceous shale near the base.

5.1.2.2 Quaternary Alluvium

Fan and stream-channel alluvium deposited by small intermittent streams covers the northernmost portion of the project area and locally extends south across U.S. Highway 6 (Figure 5–3). The unit grades vertically and laterally into the Colorado River flood plain deposit to the south and into Wasatch colluvium and sheetwash deposits to the north. Loess locally mantles older terrace gravel deposits and Wasatch bedrock above the floodplain to the north.

A uniformly thick deposit of approximately 20 to 30 ft of unconsolidated Quaternary material consisting mostly of silt, sand, and cobble gravel covers the relatively flat-lying scoured bedrock surface beneath the site. The thickness of the Quaternary material increases up to 100 ft downgradient of the site where it fills local valleys. The lateral extent of the alluvium is bounded by Wasatch outcrop to the north and by the Colorado River to the south as shown on the isopach map presented in Figure 5–6.

The lithologic character and variability of the Quaternary material was determined by examination of split barrel samples and drill cuttings obtained from the boreholes installed during the 1998 field investigation (Section 4.0). The floodplain alluvium deposited along the Colorado River consists of two units: an upper unit and a lower unit. The lower unit of the floodplain alluvium is mostly a cobble pebble gravel (Wentworth scale) with a sand matrix (60 percent gravel, 35 percent sand, 5 percent silt). Sedimentary, igneous, and metamorphic rocks derived from source areas far to the east dominate the clasts' lithology. The upper unit of the floodplain alluvium consists of silty, very fine-to medium-grained sand (80 percent) that locally contains a minor amount of pebbles and cobbles (<5 percent) and thin lenses of silt and clayey silt (15 to 20 percent). These fine-grained sands and silts along the Colorado River are probably overbank deposits. Colors are typically brown, dark brown, and dark yellowish brown (10YR Munsell notation). The borehole lithologic logs in Appendix A provide descriptions of the unconsolidated material.

5.2 Hydrologic System

Major features of the regional hydrologic system are the result of the White River uplift and the Grand Hogback located north of the site (Figure 5–1). In addition to being in the regional zone of recharge (Figure 5–2), the steeply dipping Grand Hogback monocline redirects surface drainage from the steep-walled canyon and the mountain region of the White River Plateau to the broad flat valleys of the Colorado River Basin. Many different flow systems discharge south to the Colorado River Basin. Rifle Creek drains the majority of the regional hydrologic catchment north of Rifle and is used extensively as a source of surface irrigation water in the Rifle area (Section 3.3). Other tributaries to the Colorado River that provide regional drainages north of Rifle include Government and Elk Creek. The Colorado River to the south is in the zone of regional discharge and acts as the regional ground water flow divide. Regional ground water flow is complicated by the structural, topographical, and geological variations in the Rifle area.

The major components of the local hydrologic system at the New Rifle site include the Colorado River, unsaturated alluvial sediments, saturated alluvial sediments, and saturated claystone, siltstone, and fine-grained sandstone of the Wasatch Formation. The local hydrologic system for the New Rifle site is graphically illustrated in the conceptual hydrostratigraphic model presented

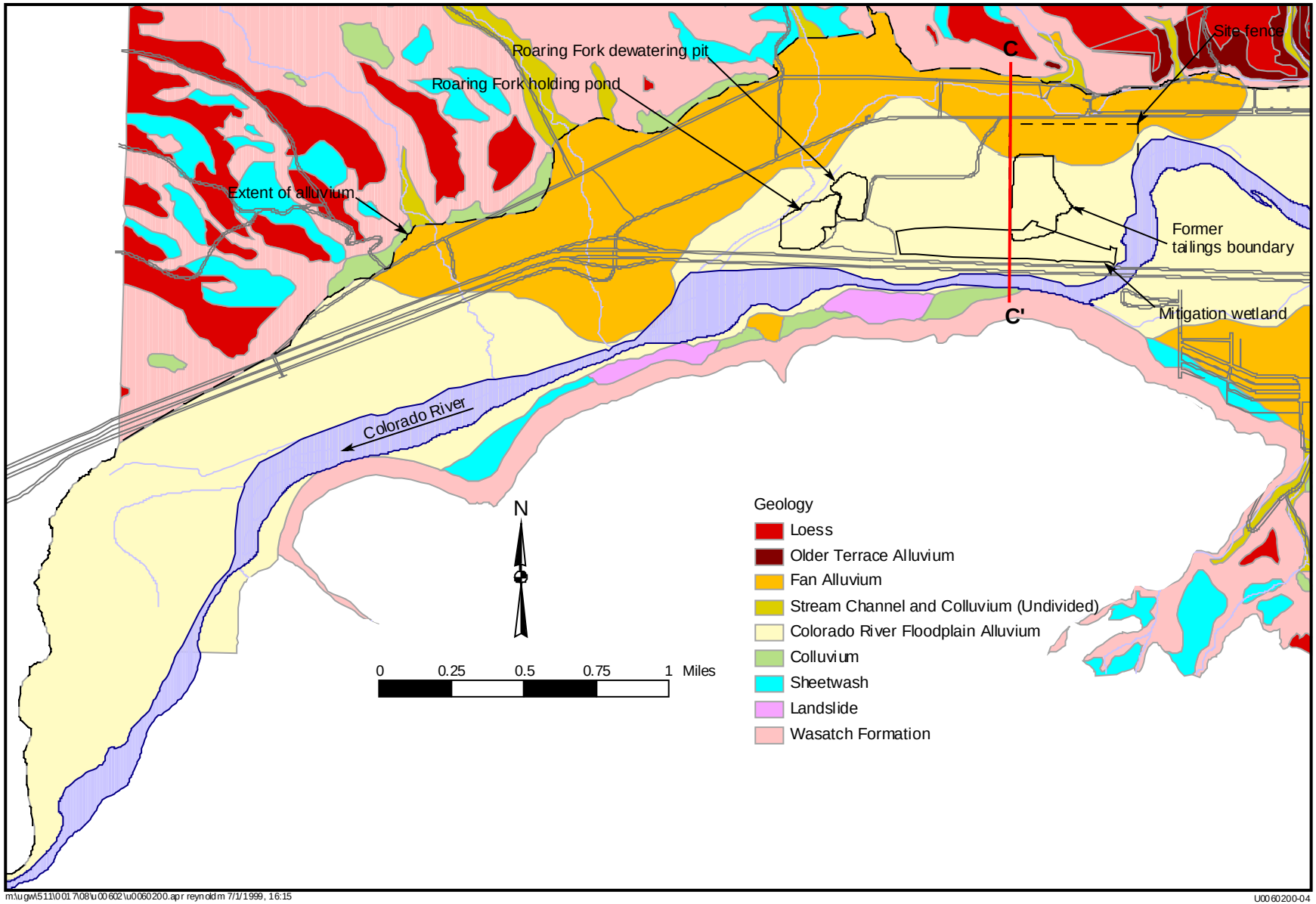


Figure 5–3. Local Surface Geology at the New Rifle Site

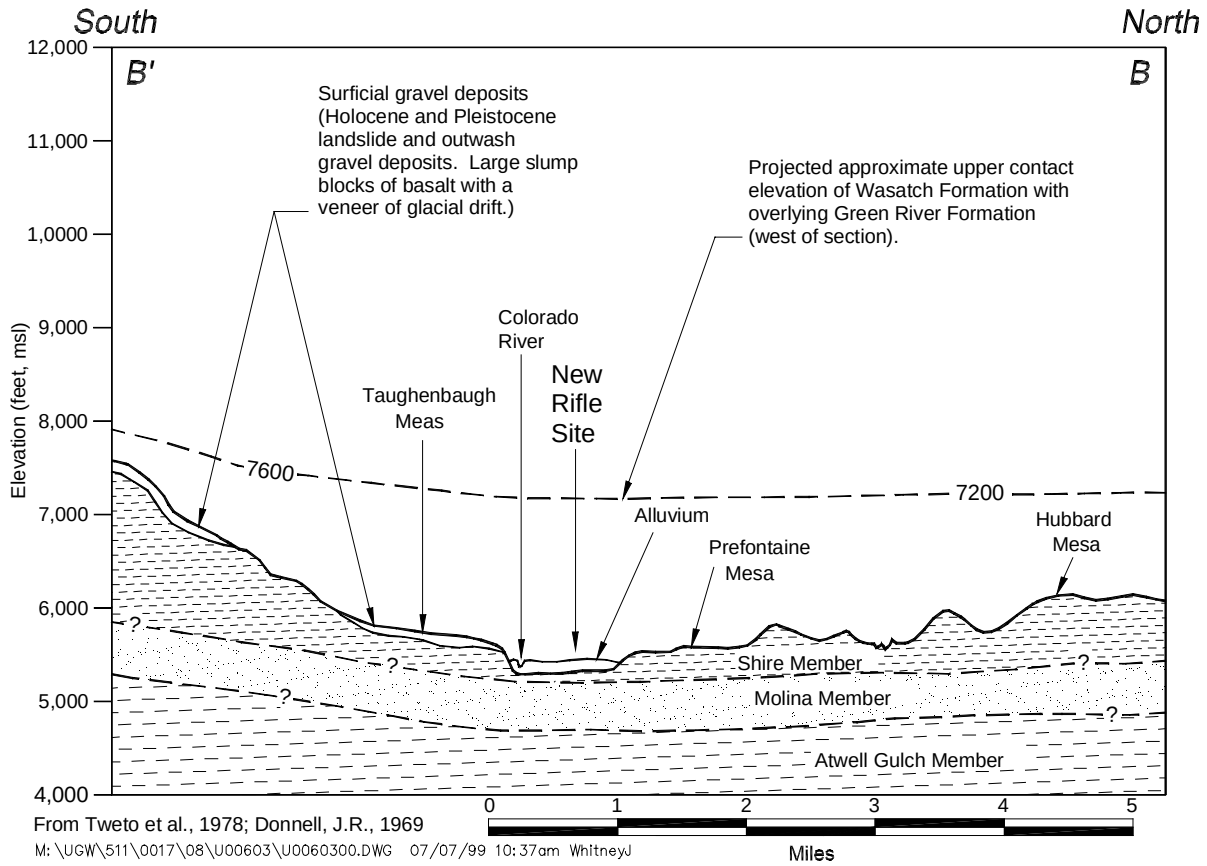


Figure 5–4. Geologic Cross-Section of the Three Wasatch Members beneath the New Rifle Site

in Figure 5–7. The saturated alluvial sediments form the unconfined alluvial aquifer. Generally, the alluvial aquifer flows in the lower unit of the floodplain alluvium, which consists of cobble pebble gravel with a sand matrix. Claystone and siltstone interbedded with minor lenticular and laterally discontinuous fine-grained sandstone are the predominant rock types of the Shire Member of the Wasatch Formation that directly underlies the unconsolidated deposits of the alluvial aquifer. The relatively low-permeability claystone beds of the Shire Member separate the overlying alluvial aquifer from the deeper and more permeable sandstone beds of the middle Molina Member of the Wasatch Formation.

5.2.1 Surface-Water Hydrology

Surface-water features at and near the New Rifle site include the Colorado River, the Roaring Fork gravel ponds, the mitigation wetland, an intermittent pond (McCauley borrow pit), the Pioneer irrigation ditch, intermittent tributary streams, and the City of Rifle wastewater treatment ponds.

The Colorado River forms the southern boundary of the New Rifle site and is the dominant surface-water feature, ultimately receiving most of the surface drainage from the New Rifle site. Precipitation falling on the site drains south directly into the river and into the mitigation wetland ponds south of the site. The river also receives ground water discharge from the alluvial aquifer along the southern portion of the site.

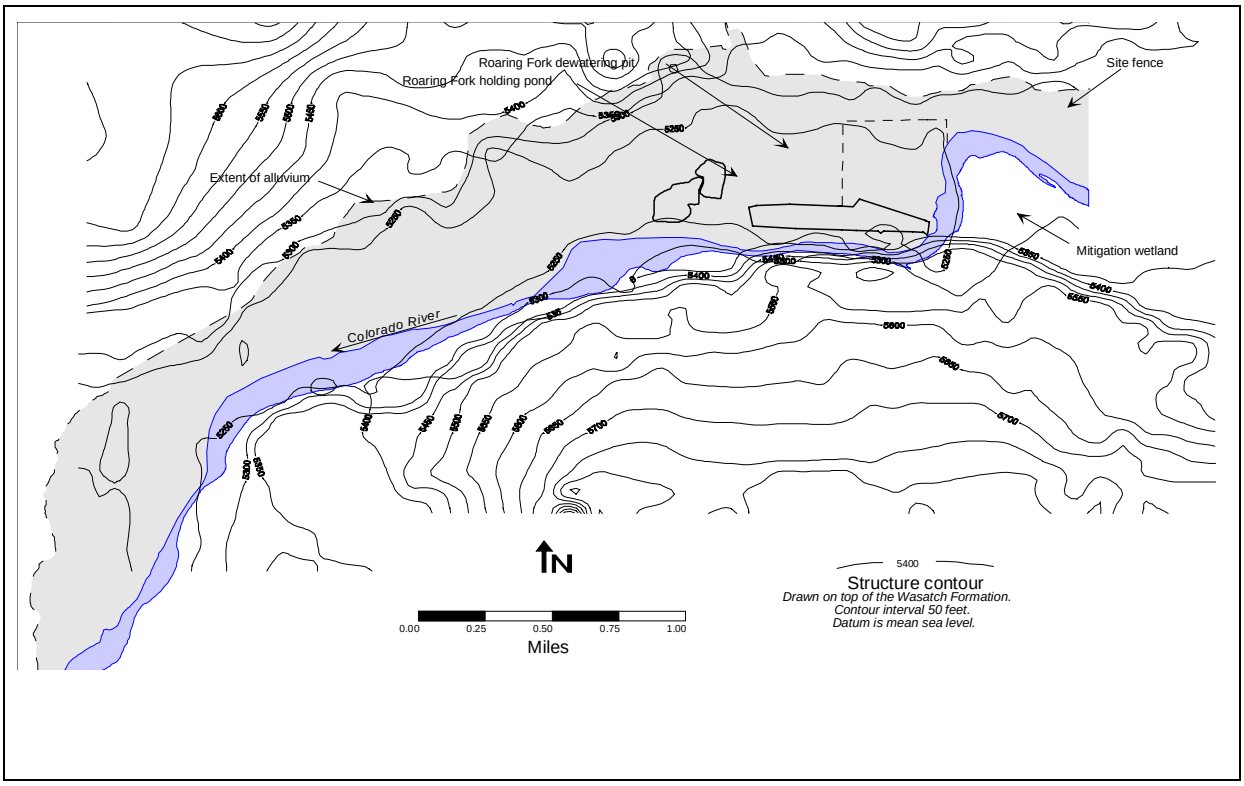


Figure 5–5. Contour Map Drawn on Top of the Wasatch Formation at the New Rifle Site

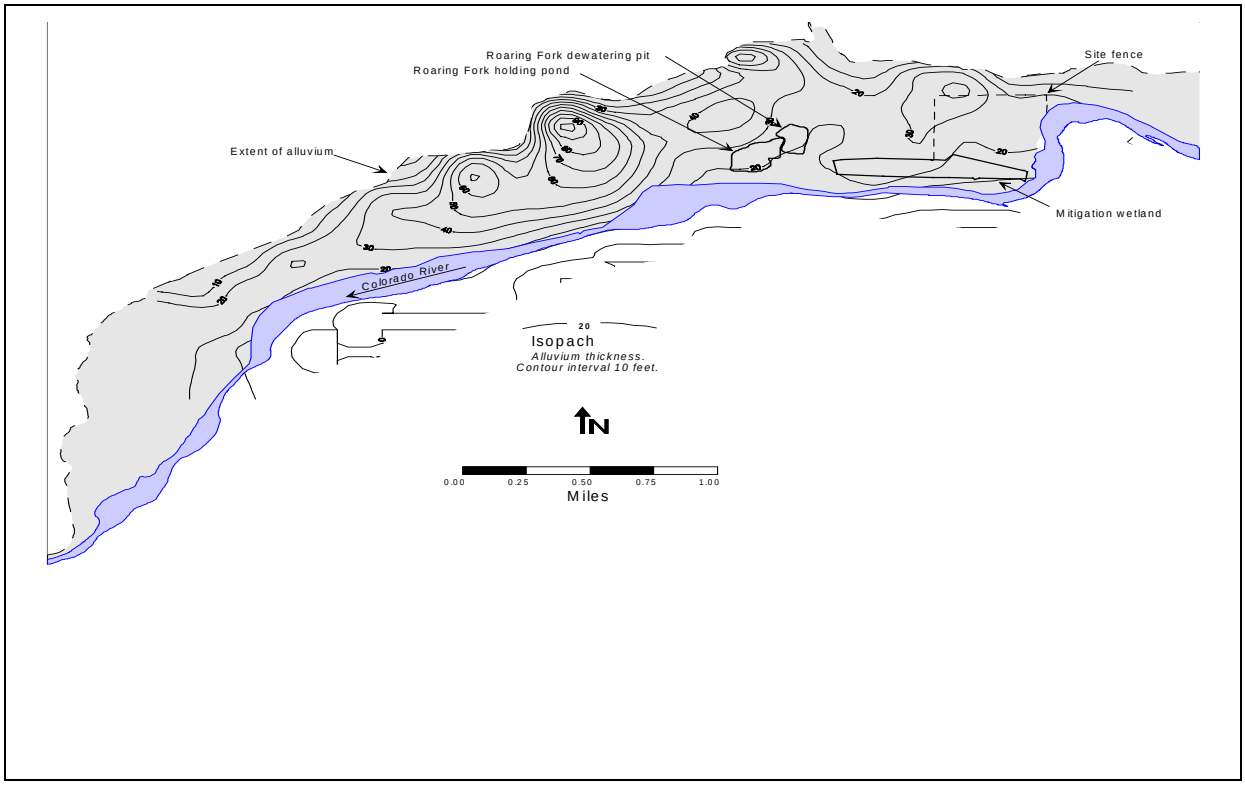


Figure 5–6. Isopach Map of the Alluvium at the New Rifle Site

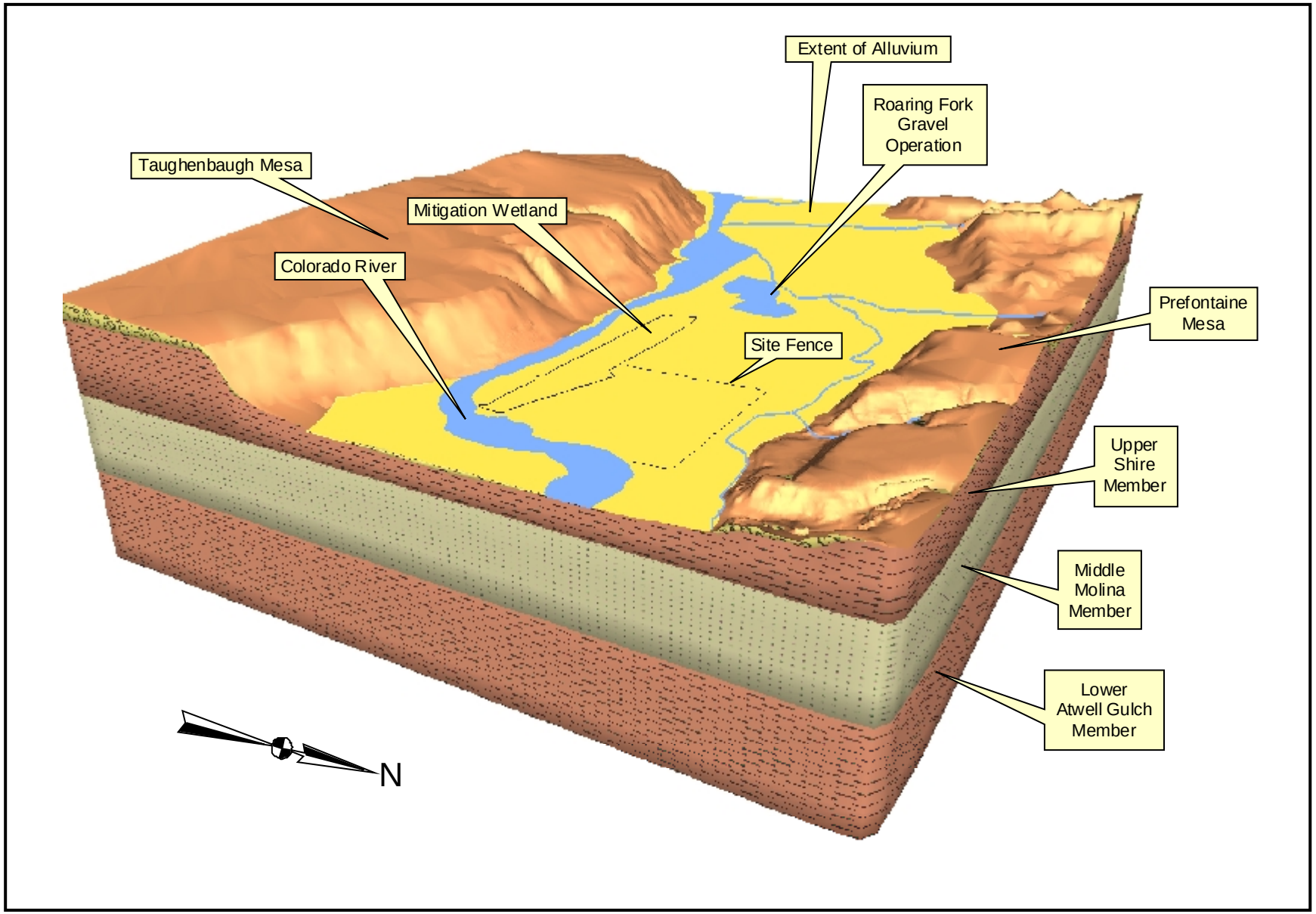


Figure 5–7. Site Conceptual Hydrostratigraphic Model

Daily averages of river-gauging measurements obtained for the Colorado River are available from the USGS. The nearest flow data are collected approximately 20 miles upstream at a location near Glenwood Springs, Colorado, and approximately 25 miles downstream at a location near the town of DeBeque, Colorado. The daily average flow rate, measured over the 20-year period from 1978 through 1997, is approximately 11 percent higher at the downstream DeBeque gauging station than at the upstream Glenwood Springs gauging station (4,060 versus 3,660 cubic feet per second [cfs]). Rifle is located approximately halfway between these two stations; therefore the average flow between the upstream and downstream gauging stations presented in Figure 5–8 is considered the best estimate of the river flow at the site. On average, the maximum discharge occurs in the spring and early summer months from the middle of May to the first of July due to snow melt. Minimum discharge occurs in the fall and winter months from October through March. Over the 20-year measurement period the average runoff was 3,848 cfs, the maximum observed was 33,800 cfs, and the minimum observed was 938 cfs.

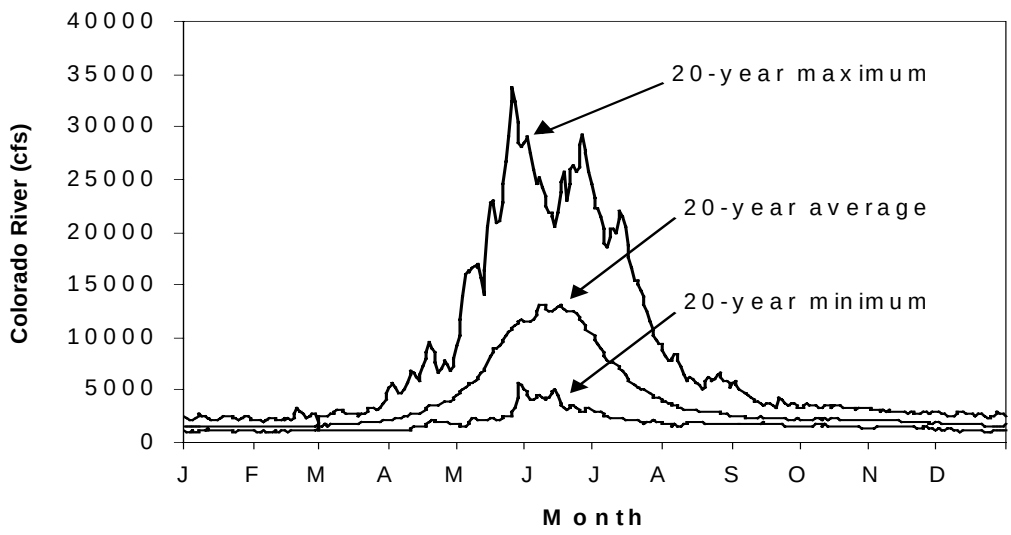


Figure 5–8. Estimated Seasonal Runoff for the Colorado River at the New Rifle Site

The mitigation wetland consists of approximately 34 acres located along the southern edge of the site, north of the Interstate 70 and the Colorado River. The wetland was constructed under a Section 404 permit issued by the U.S. Army Corps of Engineers (Permit No. 190110228) during surface remedial action to mitigate natural wetlands lost over the course of milling and remedial action activities. The mitigation wetland is designed to intersect the water table in the alluvial aquifer during the high-water period of May and June. Evaporation during high river stage causes ground water discharge from the wetlands, although this discharge is not likely to measurably alter ground water flow directions in the alluvial aquifer. During other periods of the year, the wetland acts as a ground water discharge area through the process of plant transpiration. This discharge is also likely to be insignificant.

The Roaring Fork gravel ponds are located approximately 0.5 miles west of the New Rifle site and consist of two large pits that have been excavated to bedrock. Gravel is currently being mined just east of both pits. Ground water infiltrating into the active mining area is diverted to the east pit where it is intermittently pumped into the west pit. Water pumped into the west pit has formed a perennial pond subject to evaporation.

McCauley borrow pit is located just east of the Roaring Fork gravel pits and north of the mitigation wetland. Material removed from the McCauley borrow pit was used as a source of clean sand and gravel to fill contaminated areas excavated on the former mill site during the remedial action. The bottom of the borrow pit is approximately 3 ft below ground level. Alluvial ground water rises frequently to the bottom of the pit during high-river stage and is subject to evaporation. Approximately half of McCauley pit was recently backfilled with material excavated from the reconstruction of the adjacent wetland area in May 1999 thereby reducing the amount of water lost by surface evaporation.

Pioneer ditch located north of the site and U.S. Highway 6 is an unlined irrigation ditch distributing water diverted from Rifle Creek. The ditch is constructed in the unconsolidated deposits just south of the Wasatch outcrop that forms the northern limit of the alluvial aquifer. Leakage from the ditch recharges the alluvial aquifer. The ditch flows southwest until it eventually discharges to the river just west of the Roaring Fork gravel ponds. Several measurements performed along Pioneer ditch in January 1999 indicate a range in flow between 25 and 94 gpm (Section 4.8).

Several small-unnamed intermittent tributary streams drain Webster and Prefontaine Mesas to the northwest and north of the site, respectively. Surface water flows south from the Mesas toward the Colorado River. Leakage from these intermittent streams recharges the alluvial aquifer. A flow measurement performed in January 1999 at the first unnamed tributary stream just west of the site and north of U.S. Highway 6 indicate an approximate flow rate of 8 gpm (Section 4.8).

Two wastewater treatment ponds are located east of the site in a narrow section of the alluvial aquifer bounded by Wasatch Formation outcrop to the north and the Colorado River to the south. The ponds are used as part of the City of Rifle wastewater treatment system. Any potential leakage from the ponds is not expected to significantly influence the site hydrogeology.

5.2.2 Alluvial Aquifer

The alluvial aquifer is the uppermost hydrostratigraphic unit at the New Rifle site and is composed of unconsolidated deposits that include mostly silts, sands, and gravels (Section 5.1). The unconsolidated deposits are illustrated in the geologic cross-section extending north to south through the site (Figure 5–9). Most of the alluvial ground water flows in the pebble cobble aquifer that comprises the lower unit of the floodplain alluvium. Underlying the pebble cobble aquifer are indurated claystones and siltstones that are interbedded with lenticular fine-grained sandstones of the Wasatch Formation.

The lateral extent of the alluvial aquifer at the New Rifle site is largely limited to the boundary defined by Wasatch Formation outcrops to the north and the Colorado River to the south (Figure 5–6). Approximately 4 miles to the southwest the alluvium pinches out between the river and a prominent Wasatch outcrop. Similarly, only a narrow section of the alluvium extends east of the site between the Wasatch outcrop to the north and where the course of the Colorado River abruptly turns south.

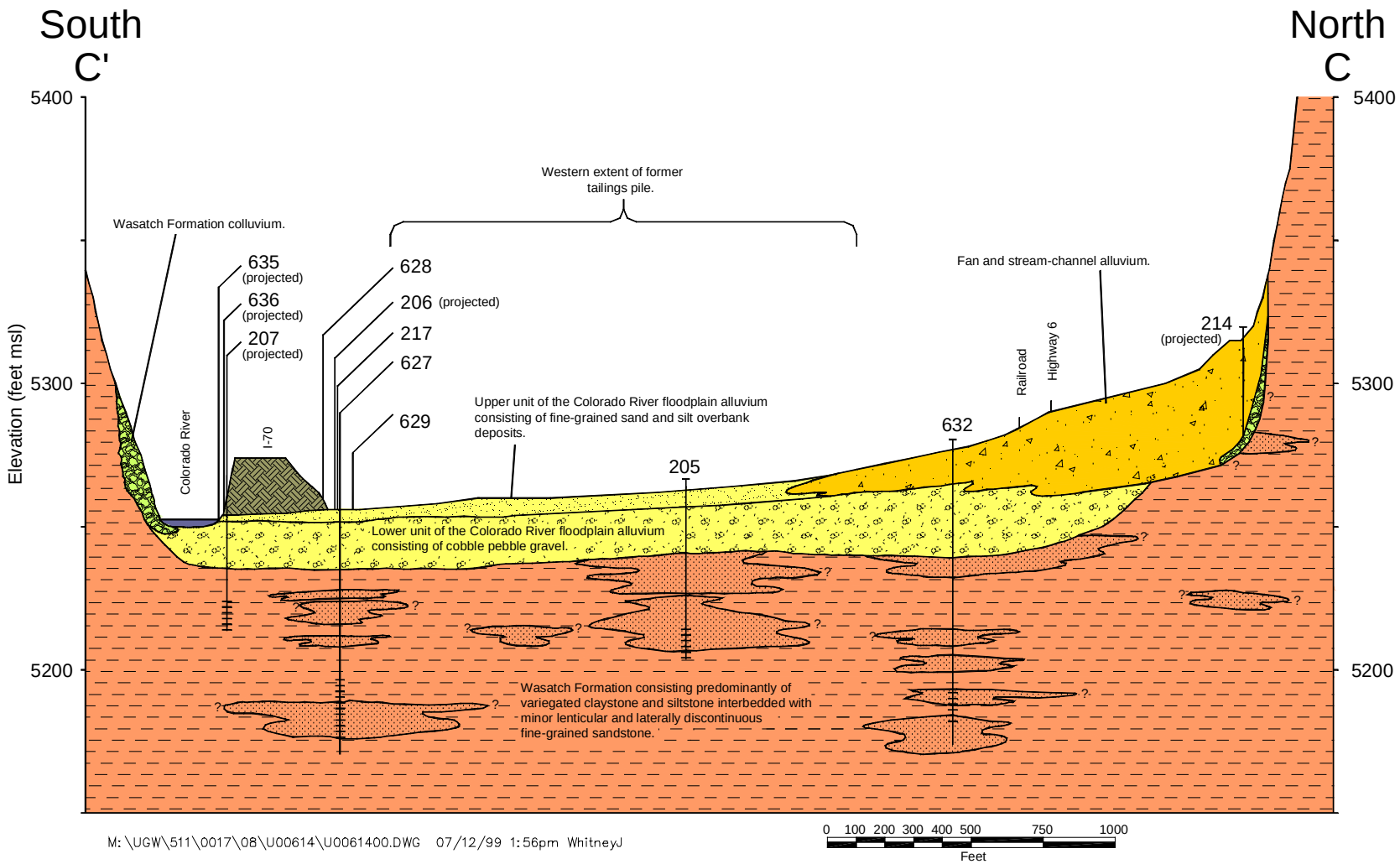


Figure 5–9. Geologic Cross Section North to South Across New Rifle Site

Alluvial deposits at the New Rifle site are approximately 20 to 30 ft thick over most of the site (Figure 5–6) with depths to ground water ranging from 5 to 10 ft. The greatest depth to water is 90 ft which occurs approximately 1½ miles downgradient of the site (RFN–170) where alluvial fan material fills a local valley. The thickness of the Quaternary material at this location is slightly over 100 ft. Saturated thickness of the aquifer over most of the project area ranges from 10 to 20 ft.

Hydraulic conductivities estimated from alluvial aquifer tests average 114 ft/day (Calculation Set U0042800). Hydraulic gradients, directed west-southwest, range from 0.0019 to 0.0040 ft/ft (Calculation Set U0060500). Average linear velocities, based on these estimates and an assumed effective porosity of 0.27, range from 0.8 to 1.7 ft/day.

Recharge to the alluvial aquifer at the site occurs mostly as infiltration of precipitation, leakage from the unnamed intermittent tributaries and Pioneer ditch north of U. S. Highway 6, and by the Colorado River, especially along the north-south reach of the river east of the site which appears to be a ground water recharge source throughout most of the year. During periods of spring runoff between May and June the Colorado River also temporarily recharges the alluvial aquifer along the southwestern portion of the site when high river flows start to exceed ground water elevations in the alluvial aquifer.

Discharge from the alluvial aquifer at the New Rifle site is to the mitigation wetland, to the Roaring Fork gravel pits, and to the Colorado River. Plant transpiration and evaporation in areas of shallow ground water depths are the only other processes by which ground water may be discharged from the alluvial aquifer. Discharge by plant evapotranspiration (ET) is considered minimal because the site is dominated by wheatgrasses (Section 4.9) which are shallow rooted and not considered to be phreatophytes (plants that root in ground water). Discharge by phreatophytes, mostly large greasewood and cottonwood communities located downgradient of the site can be significant.

5.2.2.1 Alluvial Aquifer Interactions with Surface Features

Several surface features at the New Rifle site interact with the alluvial ground water, which influence the contaminant migration from the site. Primary features of concern include the Colorado River, the Roaring Fork gravel operations, and the mitigation wetland located south of the former mill site. Understanding the interactions between these surface features provides a more complete conceptual model of the hydrologic dynamics that are required to develop the computer flow and transport simulations as part of the natural flushing alternative evaluation.

Alluvial Aquifer Interactions with the Colorado River

A stilling well was installed in the Colorado River along the east boundary of the site to measure elevation changes in the river. Electronic data loggers and pressure transducers were installed in the stilling well and in three alluvial monitoring wells located at increasing distances from the riverbank. Fluctuations in the river elevation and the associated response in the alluvial aquifer at the three monitor wells were recorded on 4-hour intervals from April 1998 through July 1999. The hydrograph presented in Figure 5–10 indicates the stages of the Colorado River fluctuate seasonally with the maximum rise occurring in the spring and early summer months from the middle of May to the first of July due to snow melt and the minimum rise occurring in the fall and winter months from October through March. The peak stage was reached on June 2, 1988,

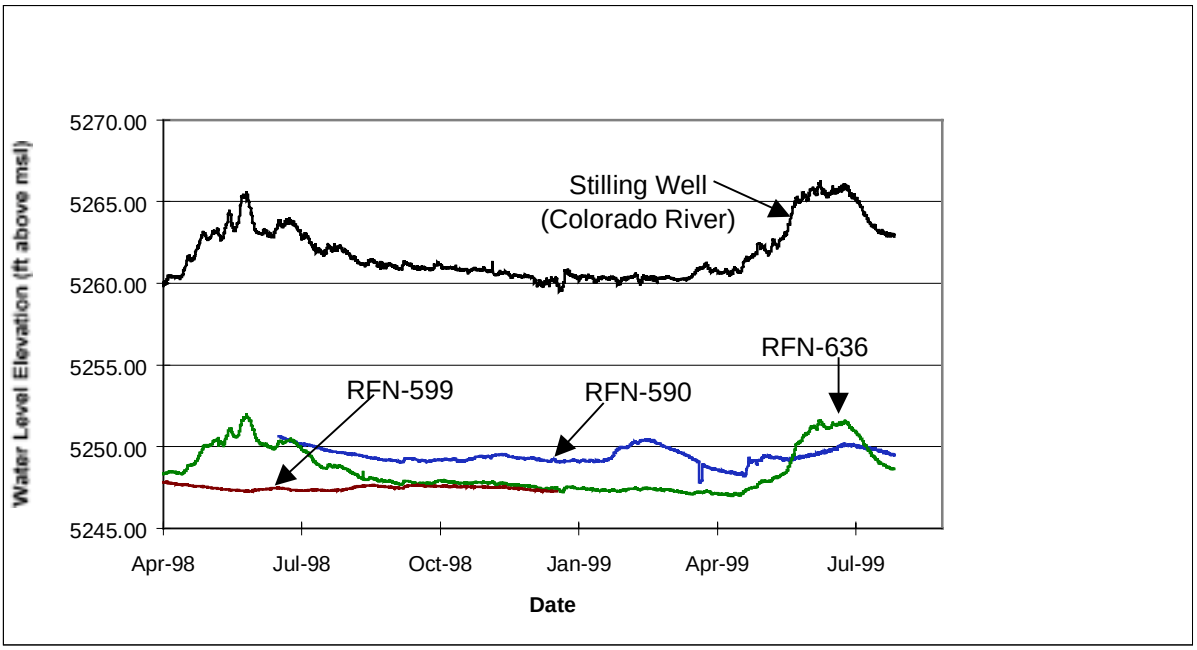


Figure 5–10. River Stage and Aquifer Response Measured at the New Rifle Site, April 1998 through July 1999

with a rise of over 5 ft and on June 10, 1999, with a rise of almost 7 ft. The lowest stage was reached on December 22, 1998. This seasonal response is consistent with the Colorado River gauging measurements obtained by the USGS over a 20-year period (Section 5.2.1.). Fluctuations in river stage produce a significant and similar response in alluvial ground water elevations near the river. This is evidenced by the hydrograph for monitor well RFN–636 which is located less than 100 ft from the riverbank. The hydrograph pattern coincides almost identically with the pattern observed for the river stilling well. Aquifer responses appear to diminish with increasing distance from the river. For example, the maximum rise in the river elevation over the measurement period shown in the hydrograph was almost 7 ft. The maximum rise in the alluvial aquifer was approximately 5 ft, 3 ft, and 1 ft, at distances of approximately 100 ft (RFN–636), 700 ft (RFN–590), and 1,800 ft (RFN–599), respectively, from the riverbank.

Alluvial Aquifer Interactions with the Roaring Fork Gravel Operations and Mitigation Wetland

The majority of ground water in the alluvial aquifer at the New Rifle site discharges during low river stage into the Colorado River along the southern boundary of the site. The southwest flow direction is shown by the January 1999 ground water elevation contours presented in Figure 5–11. As the river stage rises in the spring and early summer, the ground water maintains the southwest flow direction through the eastern portion of the mitigation wetland. Higher river elevations combined with the dewatering activities at the Roaring Fork gravel mine appears to be manipulating the natural ground water flow gradient to a near westerly direction at the west end of the mitigation wetland. Consequently, the dewatering at the Roaring Fork gravel mine tends to reduce discharge from the alluvial aquifer to the Colorado River along the southwest portion of the site during higher river levels. Ground water flow directions and perturbations in the ground water gradient associated with dewatering activities are revealed by the ground water elevation

contours presented in Figure 5–12 for July 1998. A similar contour pattern is evident in the ground water elevations for May 1999 when the river was near high stage.

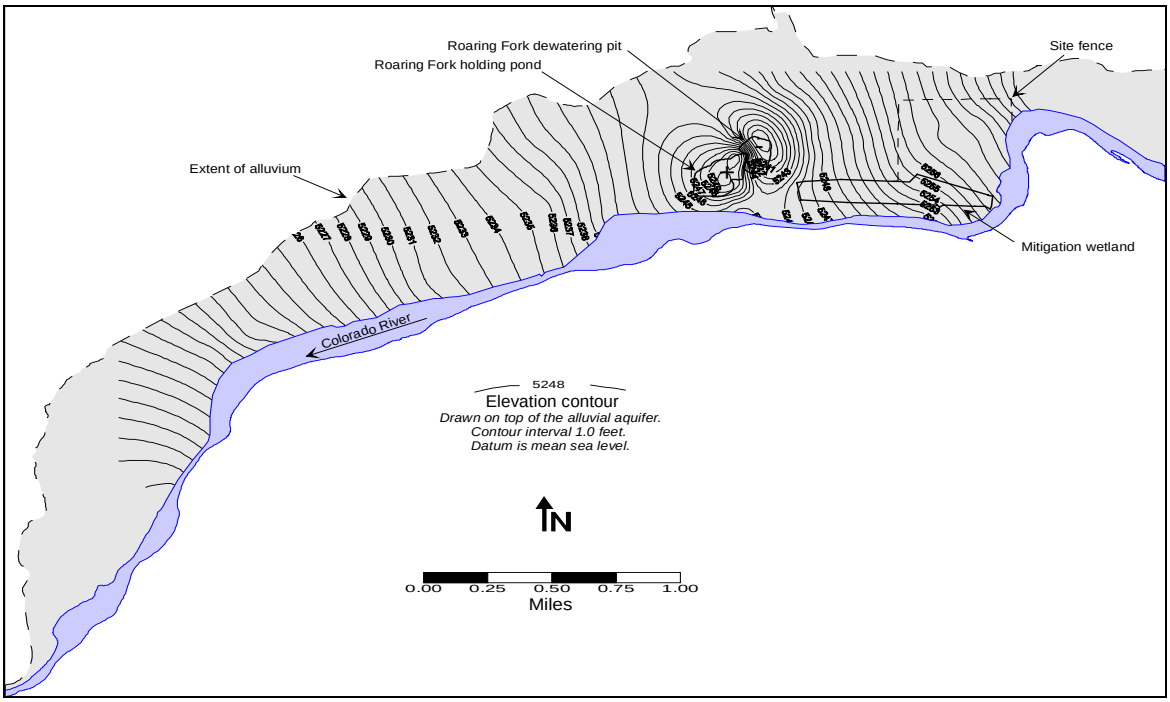


Figure 5–11. Alluvial Ground Water Elevations at the New Rifle Site—January 1999

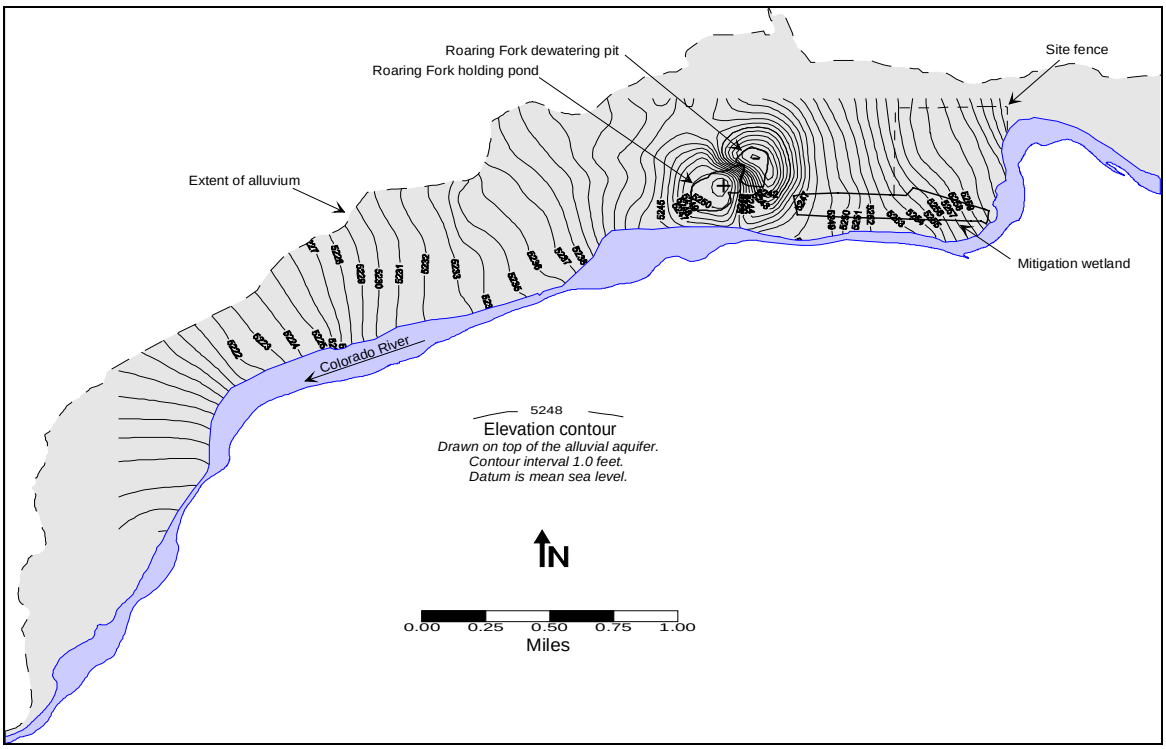


Figure 5–12. Alluvial Ground Water Elevations at the New Rifle Site During Dewatering Activities at the Roaring Fork Gravel Mine—July 1998

Dewatering at the Roaring Fork gravel mine is necessary to facilitate ongoing operations because most of the gravel deposits are located below the alluvial water table. Alluvial ground water infiltrating the active mining area is diverted to the east pit where it is pumped into the west pond and held to allow the water to evaporate or infiltrate back into the alluvial aquifer downgradient of the mining operation. In addition, some of the ground water in the holding pond ultimately recirculates back into the dewatering pit where it is then pumped again back into the holding pond. As a result, the dewatering has significantly altered the local natural ground water flow gradients. In effect, the dewatering pit is behaving as a large-diameter pumping well located next to a large diameter downgradient injection well. The pumping has effectively increased hydraulic gradients east of the dewatering pit. Such an increase is accompanied by an increase in ground water flow velocities east of the pit and a radial warping of flow directions towards the dewatering pit. Conversely, the holding pond has created a ground water mound and a radial warping of flow directions away from the west pond. A corresponding increase in ground water flow velocities west of the holding pond results due to the increase in the hydraulic gradient created from the mound. Perturbations in the local flow gradients are evidenced in the ground water elevation contour maps presented in Figures 5–11 and 5–12 for measurements collected in January 1999 and July 1998, respectively. A cone of depression in the ground water is clearly visible at the east pit. The center of the cone of depression is indicated by a relative low (i.e., -). A ground water mound is also apparent in the west pit used to hold water pumped from the dewatering pit as indicated by a relative high (i.e., +).

Potential impacts to the ground water flow gradients in the mitigation wetland due to the dewatering activities at the Roaring Fork gravel mine were evaluated using the USGS MODFLOW computer code. Details regarding the ground water flow model calibration are presented in Appendix D. Simulated ground water elevations for July 1998, modeled with no dewatering activities at the Roaring Fork mine, are presented in Figure 5–13.

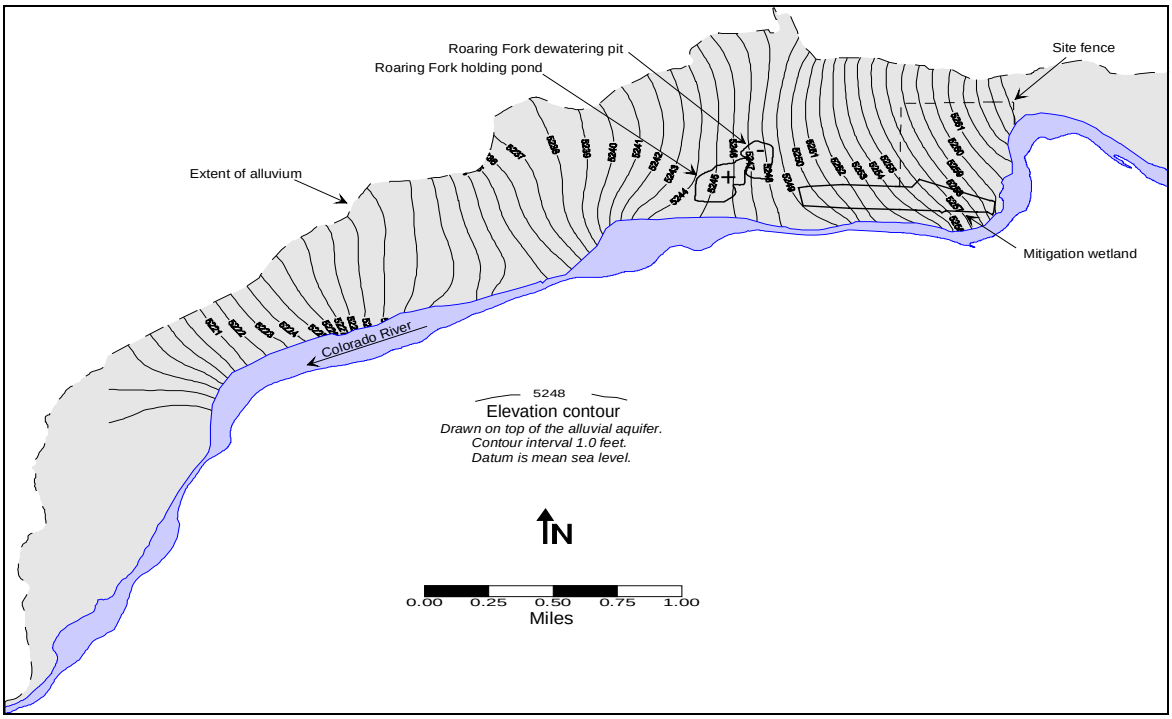


Figure 5–13. Alluvial Ground Water Elevations Predicted Using the USGS MODFLOW Model to Simulate the Natural Flow Gradient Without Dewatering Activities—July 1998

The potential impact to the natural flow gradients can be evaluated by comparing the measured July 1998 water elevations during dewatering activities to the ground water elevations simulated without dewatering. For example, the ground water elevation contour passing through the southwest corner of the wetland fence boundary during dewatering activities in July 1998 is 5,247 ft (Figure 5–12). The predicted ground water elevation contour passing through the same location with no dewatering is 5,249 ft (Figure 5–13), indicating that dewatering the Roaring Fork gravel mine is depressing the ground water elevations at the west end of the mitigation wetland by approximately 2 ft. Similar comparisons of ground water elevations indicate that dewatering has no effect on the eastern half of the wetland. These results are consistent with the observed historical decreases in ground water elevations at monitor wells RFN–589 and –590 presented in the *1998 Section 404 Monitoring Report/New Rifle Wetland* (DOE 1998a).

Vegetation mapping conducted in 1998 as part of the requirements for the 404 permit indicates visibly drier conditions at the west end of the wetland (DOE 1998a). The driest conditions were observed at the northwestern end where the ground elevation was increased during the original construction, creating a drier habitat than specified in the original design. Kochia, an upland weed, interspersed with a few desirable wetland plants and various other upland weeds, dominated this area. As a result of the depressed ground water elevations at the west end of the mitigation wetland from the dewatering at the Roaring Fork mine and the higher ground elevation than specified in the original wetland design at the northwest end of the wetland, it was determined that the hydrology was not sufficient for the establishment of wetland species as specified under the 404 permit. For these reasons, DOE reconstructed the west end of the mitigation wetland in May 1999 to restore approximately 20 acres of wetland habitat. The reconstruction lowered the surface elevation approximately 2 ft over the western half and created a small pond and oxbow to buffer impacts to the wetland plants in anticipation of changes in the ground water elevation when the Roaring Fork pumping center changes or pumping ceases and the natural ground water gradients recover, albeit these circumstances are difficult to anticipate. Details of the reconstruction design are presented in the *1998 Section 404 Monitoring Report/New Rifle Wetland* (DOE 1998a). Design drawings of the reconstructed mitigation wetland with respect to the alluvial ground water elevation, the pond, and the oxbow are presented in Figure 5–14.

5.2.3 Bedrock Aquifer

A stratified hydrogeologic unit composed of alternating layers of claystone, siltstones, and sandstones of the Shire Member of the Wasatch Formation comprise the bedrock directly beneath the site. The relatively low-permeability claystone and siltstone beds of the Shire Member form aquitards that separate the overlying alluvial aquifer from the deeper and more permeable sandstone of the middle Molina Member. The saturated sandstone beds of the middle Molina Member forms a semiconfined to confined aquifer of an undetermined thickness.

A geologic cross-section extending north to south through the site of the strata prepared from core obtained at bedrock borings RFN–207, –627, –205, and –632 is presented in Figure 5–9. Detailed lithologic descriptions of the core indicate that the bedrock immediately beneath the alluvium consists predominantly of variegated claystone and siltstone interbedded with minor lenticular and laterally discontinuous fine-grained sandstone (Appendix A). Generally the sequence fines upward. The subcrop map presented in Figure 5–15 was prepared from lithologic descriptions of bedrock samples collected at 61 boring locations. Areal characteristics of the bedrock units indicated that fine-grained sandstone is the predominant rock type that occurs at

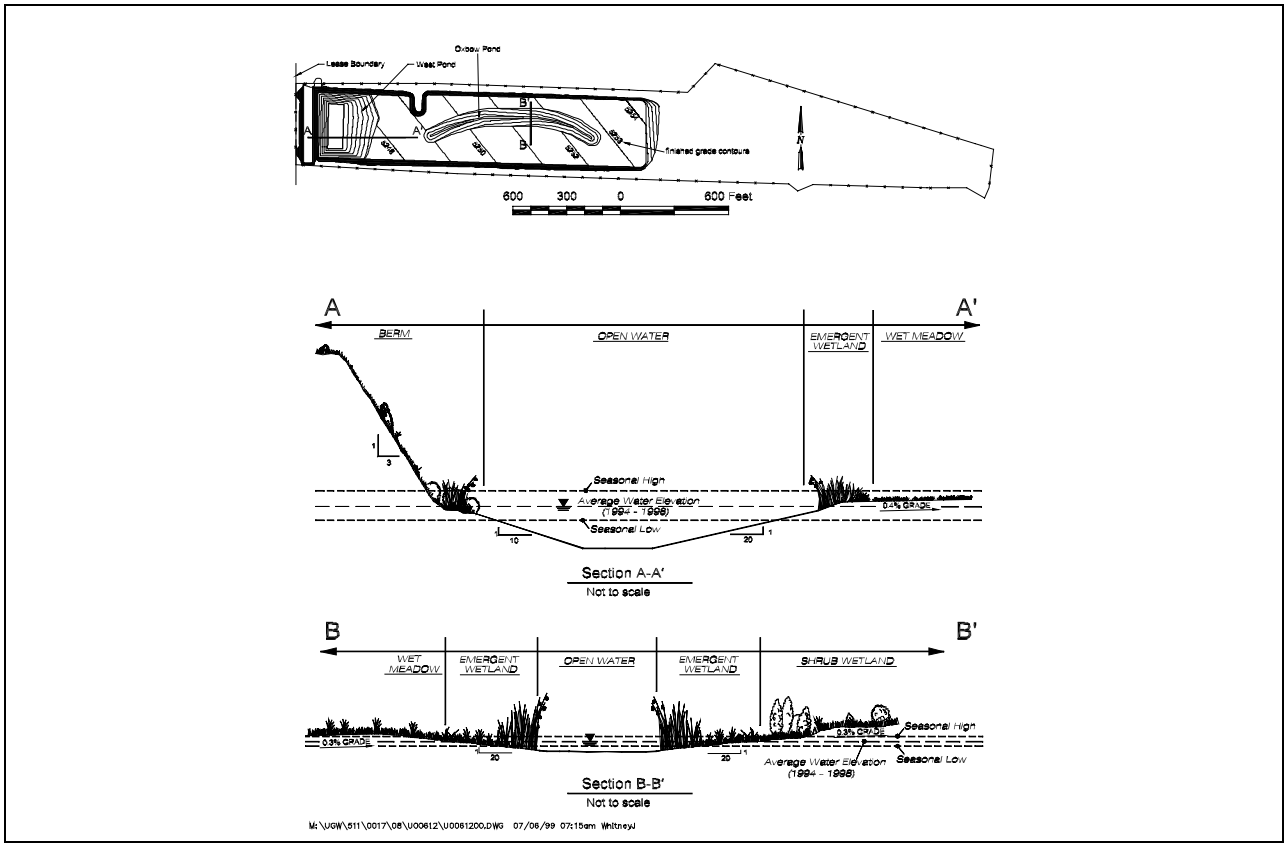


Figure 5–14. Mitigation Wetland Reconstruction Design

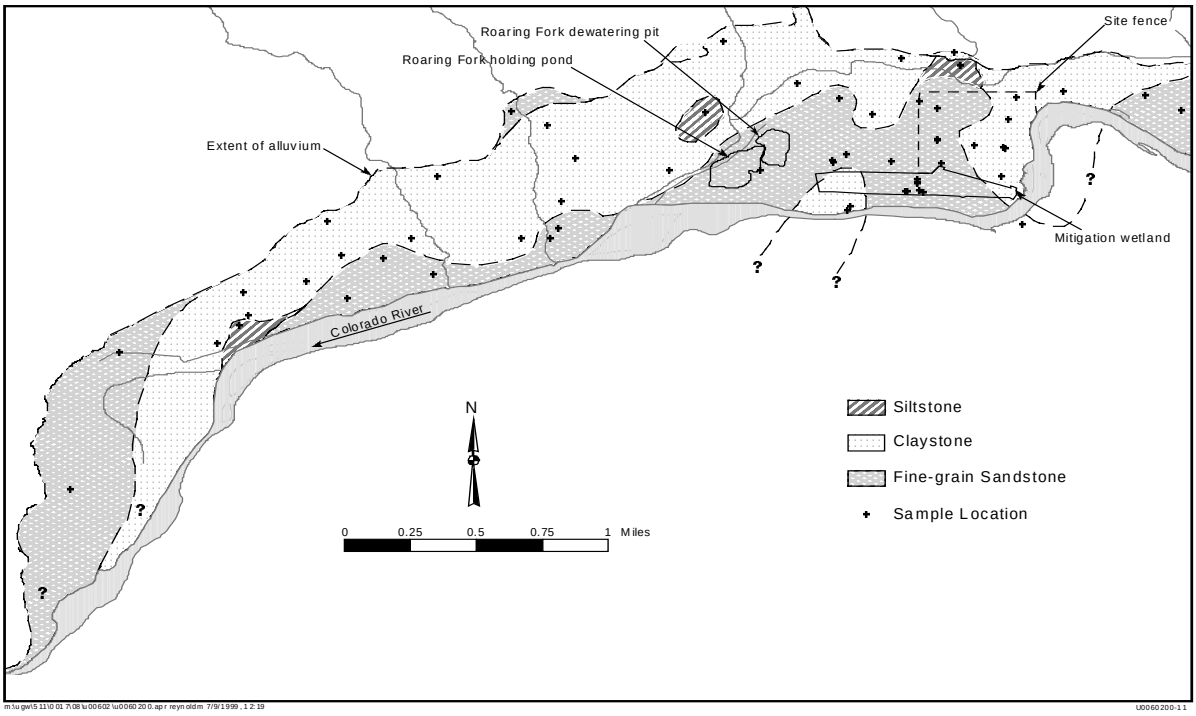


Figure 5–15. Areal Characteristics of Subsurface Bedrock Units

the contact over most of the western half of the site (Figure 5–15) and in places is present up to 14 ft thick (RFN–205). The bedrock contact beneath the eastern half of the site is dominated mostly by claystone that is also highly weathered in places. These uppermost fine-grained sandstone and claystone units at the bedrock contact appear to be hydraulically connected to and of similar hydraulic characteristics as the overlying unconsolidated sediments (Calculation Set U0042800). For this reason, the upper most beds of the Shire Member that are hydraulically connected to the alluvial aquifer are considered as part of the upper surficial flow system. The saturated sandstone beds of the deeper Molina Member that forms a semiconfined to confined aquifer of an undetermined thickness is considered the lower flow system.

Depth to water measured in monitor wells completed in the shallow portion of the Shire Member directly beneath the alluvial aquifer typically ranges from 5 to 10 ft below ground surface. In general, there are minimal vertical gradients between the alluvial aquifer and the shallow Shire Member at the bedrock contact because these units are hydraulically connected. This relationship is observed at four well pair locations where an alluvial well was completed next to a shallow bedrock well. At each paired location, the bedrock well was completed with a 10-ft screen beginning 10 ft below the contact between the alluvium and the Wasatch Formation. Alluvial monitor wells RFN–201, –636, –217, and –219 are nested with bedrock wells RFN–208, –207, –206, and –225, respectively (Plate 1). Ground water elevations in these groups of wells represent the hydraulic relationship between the alluvium and the section of the Wasatch Formation that is most susceptible to vertical contaminant invasion. Examination of the vertical hydraulic gradients reveals little or no driving force for downward contaminant migration (Calculation Set U0060500). For example, there is a slight upward vertical gradient from the Wasatch to the overlying alluvium at well pairs RFN–201/–208 (0.0001 ft/ft) and RFN–636/–207 (0.0028 ft/ft) and a slight downward gradient at well pairs RFN–219/–206 (0.0140 ft/ft) and RFN–219/–225 (0.0414 ft/ft).

Conversely, complex vertical gradients exist in the deeper portion of the Shire Member. These complex gradients are the result of aquitard forming claystone and siltstone units that separate ground water in the Shire Member from the deeper flow system in the Molina Member. These gradients are evidenced by examining ground water elevations observed in monitor well pairs that are completed at different elevations within the deeper section of the Shire Member. A comparison of water elevations measured in these deeper well-pairs indicate that some show strong upward vertical gradients, some show the reverse, and others show weak to neutral gradients. For example, a strong upward gradient is observed in monitor wells RFN–642 and –643 which are located approximately ¾ mile west of the site (Plate 1). The tops of the well screens for these wells are installed approximately 50 and 37 ft below the top of the bedrock contact, respectively. Depth to water in the deeper well (RFN–642) is consistently 60 ft higher than in the shallow well (RFN–643). The relative low water elevation in the shallow well indicates the presence of a shallow aquitard.

A reversed relationship is observed in bedrock monitor wells RFN–640 and –641 which are nested together just north of the site near the Wasatch outcrop (Plate 1). The tops of the well screens for these wells are installed approximately 120 and 60 ft below the bedrock contact, respectively. The deeper well (RFN–640) consistently shows a ground water elevation that is approximately 10 ft lower than in the shallow well (RFN–641). The relative low water elevation in the deeper well indicates the presence of a deeper aquitard.

A relative neutral gradient in the deeper Shire Member is evidenced at a nest of bedrock wells (RFN–206, –629, –627, and –628) located near the southern edge of the mitigation wetland (Plate 1). The shallower completions (RFN–206, –629 and –627) are installed with the top of the well screen approximately 6, 16, and 38 ft below the top of the bedrock contact, respectively. The deepest well (RFN–628) is installed with the top of the well screen approximately 63 ft below the bedrock contact. Ground water elevations in these wells are consistently within 1 ft of each other indicating a relative neutral gradient.

Ground water elevations measured in monitor wells completed in the shallow portion of the Shire Member indicate the potentiometric gradient is to the southwest, as evidenced by the contour map presented in Figure 5–16. Similar flow directions are observed during high and low river stage. Horizontal hydraulic gradients in the shallow Shire Member are 0.0020 ft/ft during high and low river flow (Calculation Set U0060500). Slug test measurements indicate a geometric mean of 0.017 ft/day for the hydraulic conductivity (DOE 1992, 1996d) in the Wasatch Formation. Average linear velocities, based on these estimates and an assumed effective porosity of 10 percent, range from 0.0004 to 0.0005 ft/day.

Recharge to the Wasatch Formation occurs mainly as precipitation falling directly on the outcrop to the north in the vicinity of the Grand Hogback. Discharge occurs mainly as upward leakage through the alluvial aquifer to the Colorado River.

5.3 Geochemistry

The purpose of this section is to delineate the area where contamination exists, document the individual chemical constituents that contribute to the contamination, and evaluate the fate and transport of the site-related constituents. This is accomplished by first reviewing the historical literature to identify the most probable chemicals and quantities of process water used when the vanadium and uranium milling operation was active. Results of recent field investigations are then evaluated to characterize the former source areas where the contaminants were most likely to enter the ground water flow system. The most recent information available from the 1998 field investigation (Section 4.0) is then presented in a current assessment of the surface water and ground water quality, including delineation of the nature and extent of site-related constituents occurring above natural background concentrations. Finally, computer flow and transport modeling is used to evaluate the long-term fate and transport of the site-related contaminants in the alluvial aquifer.

5.3.1 Source Areas and Contaminants

Some ground water contamination probably occurred as a result of rain water and snow melt percolating through ore stockpiled at the site. The area where potential ground water infiltration would have occurred is located at the east end of the site as designated by the former ore storage area on Figure 3–1. The primary contaminants would have been the relatively more water soluble components of the ore, such as uranium, calcium, and sulfate. The source of calcium and sulfate would have been gypsum, which was part of the ore matrix. Other ore-related contaminants include vanadium, selenium, molybdenum, and arsenic.

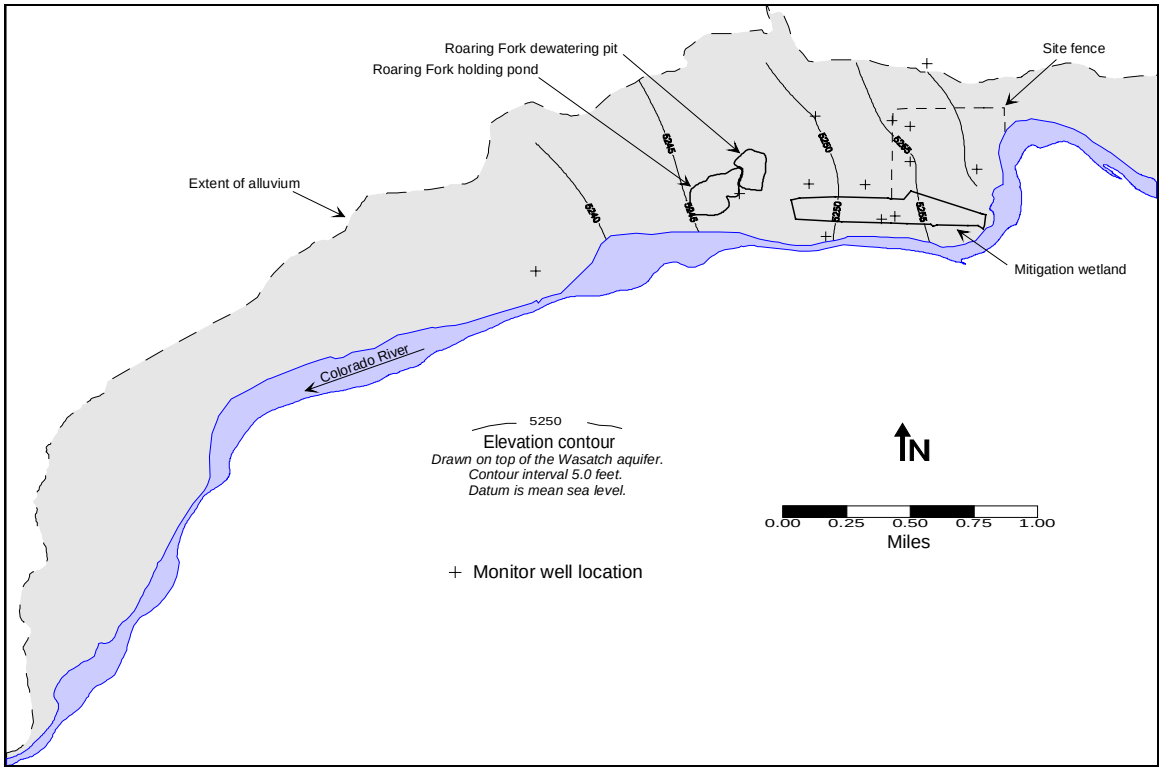


Figure 5–16. Bedrock Potentiometric Gradient

Process chemicals were an additional source of ground water contamination that probably occurred as a result of water draining from the sandy tailings that were placed as a slurry pile on the west end the site (Figure 3–1). According to Merritt (1971) the mill was built in 1958 to replace an older mill formerly located east of the town of Rifle (Section 3.2). The new mill separated uranium from low-grade vanadium ore by a direct acid leach with sulfuric acid (H_2SO_4). Ores higher in vanadium content were first treated by salt roasting with sodium chloride (NaCl). The roasted calcines were then water leached to remove soluble sodium vanadate and then the residue was acid leached to dissolve the uranium. Residues from this acid leach and from the direct acid leach circuit were washed and the recovered pregnant uranium solution was treated by solvent extraction. Vanadium-bearing raffinates were also treated by solvent extraction to recover the vanadium that was then stripped from the solvent by the addition of sulfuric acid. Ammonium chloride (NH_4Cl) was then used in a purifying step. The solvent extraction process used ethylhexylphosphoric acid (EHPA) for the recovery of both uranium and vanadium. Organic compounds used at the New Rifle site are addressed in *Phase II Organic Investigation of Ground Water Contamination at the New Rifle Site* (DOE 1997a).

Infiltration of the contaminated process water from the salt roasting and acid leaching operations would have occurred at the former mill tailings area shown in Figure 3–1. Process chemicals in the water draining from the tailings would have contributed sodium (Na^+), chloride (Cl^-), sulfate (SO_4^{2-}), and ammonium (NH_4^+) ions to the ground water. The origin of the site-related constituents that are associated with the former processing operations and the extent to which they are available to be dispersed in ground water downgradient from the site are evaluated in the following sections.

5.3.1.1 Quantity Estimates of Process Water and Chemicals

The quantity of water and process chemicals used at the New Rifle site is estimated on the basis of typical usage in uranium mills reported by Merritt (1971) and by the HEW 1962. Merritt reports water consumption rates from as low as 200 to as high as 1,000 gallons per ton of ore. The amount of chemical usage in a typical acid leach mill per ton of ore processed, based on the HEW estimate, is 30 to 500 pounds of sulfuric acid for leaching, 1 to 30 pounds of ammonia and 10 to 15 pounds of sodium carbonate to adjust the pH, 1 to 3 pounds of sodium chlorate as an oxidant, and 50 to 160 pounds of sodium chloride for roasting vanadium ores.

5.3.1.2 Former Tailing Pile Contamination

Limited information is available on the chemical composition of the contaminated process water and pore fluids that could have infiltrated to the ground water from the former tailings and ore stockpile areas. However, an estimate of contaminant concentrations in the tailings pore fluids can be obtained from leach studies of the tailing material. Water leaching tests from tailing core samples obtained from the New Rifle site were performed by Markos and Bush (1983). Average concentrations for selected contaminants (mass of analyte extracted per mass of tailings) are presented in Table 5–1. Also included in Table 5–1 are results of water leachate of soil core samples collected at selected background locations located between the eastern edge of the site and the Colorado River. Surface samples collected at these background locations were eliminated from the average to minimize any bias that may be introduced from windblown tailings since remediation was not yet completed at the time they were collected.

Table 5–1. Average Concentration of Selected Contaminants in Leachate from Tailings and Background Soils

Constituent	New Rifle Tailings Leachate (mg/Kg)	Background Soil Leachate (mg/Kg)
Arsenic	1.7	0.5
Chloride	81.2	76.0
Manganese	4.1	0.4
Molybdenum	1.4	0.5
Selenium	3.6	0.4
Sulfate	11,226	303
Uranium	0.3	0.3
Vanadium	19.6	44.4
Number of samples	37	7

Average concentrations of all the selected constituents in the tailing core samples collected at the New Rifle site, with the exception of uranium and vanadium, are elevated with respect to the average background values. The relatively low uranium and vanadium leachate concentrations in the tailing material reflect the efficient extraction process used at the New Rifle site and suggest that a significant amount of the uranium and vanadium detected in the ground water may be from process fluids discharged to the evaporation ponds.

These elevated constituents in the leachate represent water-soluble contaminants that could have entered the surficial ground water system as process water, precipitation, or irrigation runoff that

leached through the tailings pile. Thus, these are the constituents most likely to be present in the subpile soils and dispersed in the ground water downgradient from the former source areas.

5.3.1.3 Subpile Soil Contamination

The tailings pile and residuals in the soils that exceeded 15 pCi/g radium-226 were completely removed from the site by 1996 as part of the UMTRA Surface Project. However, site-related inorganic constituents may have leached into the soils below the former tailings pile and gone undetected during the radiometric assessment for the tailings removal. For this reason, samples of the soils directly beneath the former tailings footprint and near the center of the former evaporation pond were collected and analyzed for arsenic, molybdenum, selenium, uranium, and vanadium (Section 4.5) to determine if these areas are likely to be continuing sources of ground water contamination. All of these constituents were identified in the BLRA (DOE 1996a) as the most significant site-related constituents occurring in the alluvial aquifer.

Three background and three on-site locations were sampled (Figure 4–5). Selected soil samples were subjected to two sequential leachings in the laboratory. Each leach represents a scenario that might cause the mobilization of contaminants from soils into the ground water. The first leach used an uncontaminated synthetic ground water to represent the effects that a high water table might have if it were to contact contaminated soils. Concentrations of selected constituents extracted from background and subpile soils using the synthetic ground water leach are summarized in Table 5–2 and presented in Figure 5–17.

Table 5–2. Summary of Constituents Extracted by Synthetic Ground Water from Background and Subpile Soils

Monitor Well	Area	Constituent (mg/Kg)				
		As	Mo	Se	U	V
292	Background	0.31	0.12	<0.2	0.38	0.5
658	Background	0.26	0.1	<0.2	0.1	0.5
659	Background	0.22	<0.1	<0.2	<0.1	<0.1
216[a]	Evaporation Pond 11 ft	11.9	16.3	0.9	19.1	298
216[a]	Evaporation Pond 7 ft	0.4	1.1	<0.2	0.7	7.7
225	North Tailing Pile	0.35	1.5	<0.2	0.56	1.1
218[a]	South Tailing Pile	0.3	1.7	<0.2	0.13	3.2

[a]Average of samples collected from same depth

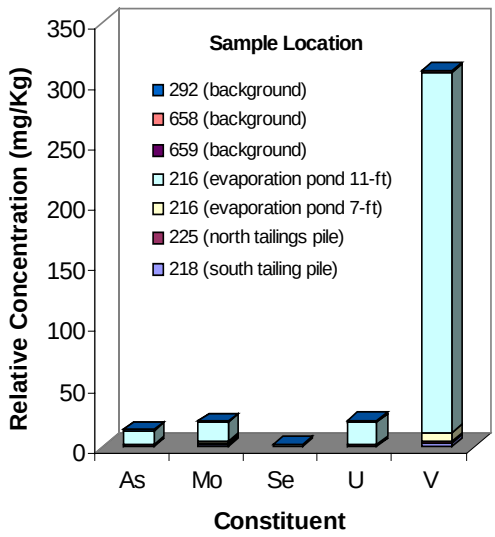


Figure 5–17. Comparison of Constituents Extracted by Synthetic Ground Water from Background and Subpile Soils

The second leach consisted of a 5 percent nitric acid solution that will remove carbonate minerals and iron and manganese oxyhydroxides. These phases are believed to be the main metal and uranium scavengers in the soils. Although it is not likely that water of this acidity would ever contact the soils, the removal of oxyhydroxides might occur if land uses changed significantly (for example, agricultural use could cause changes in redox conditions that would influence mineral dissolution). The 5 percent nitric acid is considered a worst-case scenario. Concentrations of selected constituents extracted from background and subpile soils using the 5 percent nitric acid leach are summarized in Table 5–3 and presented in Figure 5–18.

Table 5–3. Summary of Constituents Extracted by 5 percent Nitric Acid Solution from Background and Subpile Soils

Monitor Well	Area	Constituent (mg/Kg)				
		As	Mo	Se	U	V
292	Background	1.4	0.27	<0.2	0.67	4.3
658	Background	0.99	0.41	<0.2	0.27	1.7
659	Background	0.82	<0.1	<0.2	0.16	1.5
216[a]	Evaporation Pond 11 ft	164.5	20.6	2.3	48.9	470.5
216[a]	Evaporation Pond 7 ft	2.7	3.3	<0.2	1.5	43.5
225	North Tailing Pile	2.4	4.4	<0.2	1.1	37
218[a]	South Tailing Pile	1.6	2.9	<0.2	0.8	23

[a]Average of samples collected from same depth

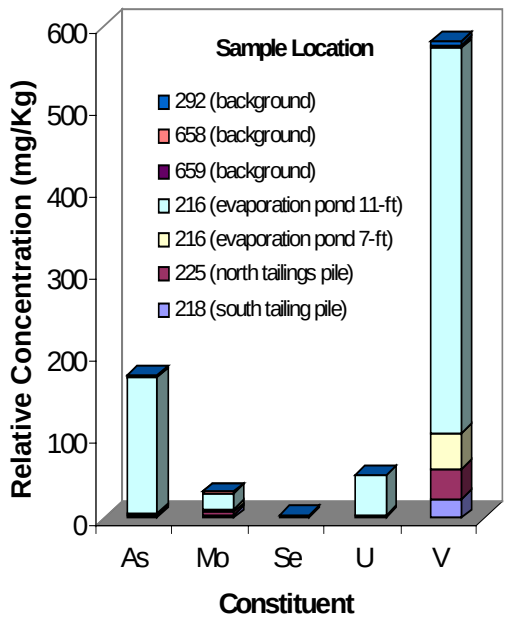


Figure 5–18. Comparison of Constituents Extracted by 5 percent Nitric Acid Solution from Background and Subpile Soils

The sum of the synthetic ground water and nitric acid leach results obtained for each sample are summarized in Table 5–4. These results provide an estimate of the total amount of leachable contaminants per mass of soil. For comparison, concentrations observed in background soils and sediments for the western United States are also presented (Shacklette and Boerngen 1984). Results obtained for the subpile soil samples are evaluated in combination with background soil results to qualitatively evaluate the likelihood that the subpile soils present a continuing contaminant source. Results for each site-related constituent are discussed separately below.

Table 5–4. Comparison of the Total Amount Extracted from Site-Specific Background and Subpile Soil Samples to the Average Background Concentrations in Soils and Sediments for the Western United States

Monitor Well	Area	Constituent (mg/Kg)				
		As	Mo	Se [a]	U	V
292	Background	1.7	0.4	0.2	1.1	4.8
658	Background	1.3	0.5	0.2	0.4	2.2
659	Background	1.0	0.1	0.2	0.3	1.6
216[b]	Evaporation Pond 11 ft	176.4	36.9	3.2	68.0	768.5
216[b]	Evaporation Pond 7 ft	3.1	4.4	0.2	2.2	51.2
225	North Tailing Pile	2.8	5.9	0.2	1.7	37.6
218[b]	South Tailing Pile	1.9	4.6	0.2	0.9	26.0
Average for the Western U.S.		5.5	0.9	0.2	2.5	70

[a]One-half the detection limit used to calculate the total for the site-specific locations
[b]Average of samples collected from same depth

Arsenic in Subpile Soil

The amount of arsenic leached by the synthetic ground water from subpile soil samples collected at the former tailing piles and from the evaporation pond sample collected at 7 ft below ground surface ranges from 0.3 to 0.4 mg/Kg (Table5– 2). These values are very similar to the relatively low arsenic concentrations of 0.22 to 0.31 mg/Kg leached from the background soil samples. Arsenic concentrations significantly exceed background levels only for the sample collected 11 ft below ground level near the center of the former evaporation pond which received processing wastes, therefore, it is expected that the highest concentrations would be at this location.

Similarly, the amount of arsenic leached from the subpile soils by the relatively harsh 5 percent nitric acid solution (Table 5–3) is also very similar to the amount leached from the background soil samples at all locations with the exception of the 11-ft sample collected near center of the former evaporation pond. At this location, the leachate concentration of 164.5 mg/Kg is significantly higher than the upper range in natural background (1.4 mg/Kg obtained at location 292).

Overall, with the exception of the 11-ft sample collected at the former evaporation pond, the extractable arsenic amounts are relatively low, especially for the synthetic ground water leach, and thus are probably not a significant contributor as a continuing source of ground water contamination. Arsenic concentrations in subpile soils above natural background at location 216 (11 ft) appear to be limited in the vertical and horizontal extent. The subpile soil at this location could potentially provide a continuing source of arsenic in the ground water, however, because the extent is limited and the relatively high K_d (Section 4.4) favors partitioning to soil, it is unlikely that any significant amount of arsenic would be contributed to the ground water.

Molybdenum in Subpile Soil

Relatively low amounts of molybdenum were leached from most of the subpile soil samples by the synthetic ground water solution and by the nitric acid. All the results are less than 5.0 mg/Kg with the notable exception of the 11-ft sample collected at the former evaporation pond. The K_d determined for molybdenum is relatively low and favors partitioning to ground water (Section 4.4). The relatively high mobility indicated by the low K_d value suggests that most of the molybdenum has already been leached from the subpile soils with the exception of isolated pockets such as location 216 (11-ft depth). Therefore, it is unlikely that residual molybdenum from any isolated pockets will provide a significant long-term contribution to the ground water at the New Rifle site.

Selenium in Subpile Soil

Leachate concentrations obtained from the background soil samples are less than the 2 µg/L detection limit (which corresponds to 0.2 mg/Kg in subpile soils) for all the selenium analyses. This detection limit is consistent with the average concentration of 0.2 mg/Kg selenium observed in background soils and sediments for the western United States. All the subpile soil leachate analyses, with the exception of the 11-ft sample collected at location 216, are also less than the detection limit indicating that residual selenium is not present in the subpile soil at these locations. Concentrations at location 216 (11 ft) are also low, only marginally exceeding the natural range in background, suggesting the subpile soils do not present a continuing source of ground water contamination.

Uranium in Subpile Soil

Most of the uranium amounts extracted from the subpile soil samples by the synthetic ground water solution and by the nitric acid are relatively low. In all cases, with the exception of the 11-ft sample collected at location 216, the total amount of uranium extracted is less than the 2.5 mg/Kg observed in background soils and sediments for the western United States.

The mobility for uranium is relatively high as indicated by the low K_d (Section 4.4). The relatively high mobility suggests that most of the uranium has already been leached from the subpile soils with the exception of small isolated pockets such as location 216 (11-ft depth). Uranium concentrations in the subpile soils at location 216 (11 ft) appear to be limited in the vertical and horizontal extent. Therefore, it is unlikely that residual uranium from any isolated pockets will provide a significant long-term contribution to the ground water at the New Rifle site.

Vanadium

Vanadium amounts extracted by the synthetic ground water solution and by the nitric acid from sample 216 (11 ft) are significantly greater than for all other subpile soil samples. This is clearly evident by the data presented graphically in Figures 5–17 and 5–18. In all other cases the total amount of vanadium extracted is less than the 70 mg/Kg observed in background soils and sediments for the western United States (Table5– 4).

Elevated vanadium concentrations in the subpile soil at location 216 (11 ft) appear to be limited in the vertical and horizontal extent. Location 216 is near the northeast corner of the mitigation wetland and the elevated vanadium could be associated with an organic-rich zone, which would enhance contaminant adsorption. The borehole lithologic log for this location indicates a strong organic odor near the depth the sample was collected (Appendix A). The subpile soil at location 216 could potentially provide a continuing source of vanadium in the ground water, however, because the extent is limited and the relatively high K_d (Section 4.4) favors partitioning to soil, it is unlikely that any significant amount of vanadium would be contributed to the ground water. Relatively low vanadium concentrations observed in the ground water at this location support this hypothesis (Appendix C).

5.3.2 Surface Water Quality

Permanent surface water features at the site include the Colorado River, the mitigation wetland, and the Roaring Fork gravel ponds. Contaminated alluvial ground water from the site has the potential to discharge into these three types of surface waters, all of which are located downgradient from the former source areas. Impacts to these surface waters are evaluated by comparison of water quality results obtained from samples collected at similar surface water features that are unaffected by site-related contamination.

Background surface water quality for samples collected February 1998 through January 1999 are summarized in Table 5–5 and presented in entirety in Appendix B. Concentrations for most constituents in the background pond samples are generally higher than for river samples. Water tends to evaporate from standing water bodies in the semiarid climate of western Colorado. Thus, the relatively higher sulfate, TDS, and trace metals levels observed in the pond samples than

observed in the river water samples can probably be attributed to concentration through evaporation.

Table 5–5. Background Surface Water Quality

Analyte[a]	Unit	Background Gravel Pond			Background One-Mile Pond[c]			Background River		
		FOD	Mean[b]	Range	FOD	Mean[b]	Range	FOD	Mean[b]	Range
Major										
Ammonia as NH_4	mg/L	4/4	.02345	.0094-.0468	8/8	.05666	.025-.111	6/8	.04472	.0082-.131
Calcium	mg/L	4/4	72.63	54.6-84.5	3/3	91	77-112	8/8	64.05	37-71.2
Magnesium	mg/L	4/4	75.15	69.5-80.7	3/3	63.5	42.5-91.1	8/8	15.48	7.67-24.7
Nitrate	mg/L	1/4		<.035-.0719	1/8		<.0406-3.03	6/8	.4153	<.011-1.05
Sodium	mg/L	4/4	398.5	355-435	3/3	92.4	63.2-113	8/8	106	25.1-180
Sulfate	mg/L	4/4	674.8	594-748	8/8	312.8	249-538	8/8	123.4	39.8-224
Metal										
Antimony	mg/L	0/4		<.001-<.001	0/3		<.001-<.001	0/8		<.001-<.001
Arsenic	mg/L	0/4		<.001-<.001	5/8	.00212	<.001-.0052	0/8		<.0001-<.001
Cadmium	mg/L	0/4		<.001-<.001	0/8		<.001-<.001	0/8		<.001-<.001
Iron	mg/L	0/4		<.003-<.005	7/8	.05005	.0096-.111	2/8		<.003-.0321
Lead	mg/L	0/4		<.001-<.001	0/3		<.001-<.001	0/8		<.001-<.001
Manganese	mg/L	4/4	.02903	.0015-.108	8/8	.2363	.0341-.695	8/8	.05021	.0059-.343
Molybdenum	mg/L	4/4	.02178	.0193-.0241	7/8	.003663	.001-.011	7/8	.00727	.0011-.0098
Selenium	mg/L	2/4	.00085	<.001-.0012	4/8	.002427	.0019-.007	2/8		.00038-.001
Uranium	mg/L	4/4	.02638	.023-.0301	8/8	.01387	.0047-.0293	7/8	.00265	<.001-.0079
Vanadium	mg/L	0/4		<.001-<.0079	4/8	.004127	.0018-.0104	0/8		<.001-<.0072
Other										
Alkalinity as $CaCO_3$	mg/L	4/4	427.5	354-475	5/5	201.7	109-322	8/8	137	93-194
Fluoride	mg/L	2/2	.8095	.803-.816	7/7	.3106	.238-.408	3/3	.2285	.129-.264
Redox Potential	mV	4/4	198.5	164-214	3/3	91	20-175	8/8	151.7	56-271
Specific Conductance	umhos /cm	4/4	2515	2340-2960	5/5	983.8	600-1585	8/8	2123	884-8720
Sulfate/Chloride		4/4	3.501	3.293-3.685	5/5	7.258	.6952-29.24	8/8	.8646	.7193-1.231
Total Dissolved Solids	mg/L	3/3	1677	1510-1880	2/2	996	912-1080	7/7	613.6	520-885
pH	s.u.	4/4	8.248	7.84-8.64	5/5	8.085	7.22-8.44	8/8	8.626	7.52-9.01
Radioactive										
Gross Alpha	pCi/L	3/4	21.48	12.79-37.86	3/3	13.14	6.95-18.97	0/8		<2.26-<10.6
Gross Beta	pCi/L	1/4		<11.18-25.45	1/3		<5.49-19.53	1/8		3.1-12.56
Lead-210	pCi/L	0/4		<.91-<1.37	0/3		<.89-<1.31	0/8		<.89-<1.33
Polonium-210	pCi/L	0/4		<.06-<.16	0/3		<.08-<.37	0/8		<.05-<.24
Radium-226	pCi/L	1/4		<.1-.84	3/3	.2833	.18-.47	6/8	.1622	<.09-.25
Radium-228	pCi/L	0/4		<.63-<.92	0/3		<.6-<.95	0/8		<.42-<1.03
Thorium-230	pCi/L	0/4		<.8-<1.6	0/3		<.8-<1.6	0/8		<.8-<1.6

[a]All measurements are filtered except specific conductance, pH, redox potential, and alkalinity as $CaCO_3$ (which are unfiltered).
[b]One-half the detection limit was used to calculate the mean; mean was not calculated if fewer than half of the samples were above detection.
[c]Two 1993 samplings of location 0539 included in the One-Mile Pond group. The five One-Mile Pond wetland locations (1201–1205) were pooled as one location to calculate the mean.

5.3.2.1 Background Surface Water Quality

Background water quality is defined as the quality the water would have if uranium-milling activities had not taken place. Background surface water quality was determined by evaluating analytical results of water samples collected at selected locations upgradient and up river from the site. Background sample locations were established at two river locations (538 and 545), seven wetland locations at One Mile Pond (539, 570, and 1201–1205), and one active gravel mining pond located near the Colorado River (580). These surface water sample locations are shown on Figure 5–19.

5.3.2.2 Site Impacts on Surface Water

The Colorado River, mitigation wetland, and the Roaring Fork gravel pond are the only permanent surface water present at the site. These surface water features all receive discharge from the alluvial aquifer. River samples were collected adjacent to and downstream of the site at locations 574 and 548, respectively, to determine impacts on the river, if any, from site contamination. Samples were also collected from the mitigation wetland (1206–1210) and from the Roaring Fork gravel pond (1211–1215 and 575). Sample locations are shown in Figure 5–19. Surface water quality results for samples collected February 1998 through January 1999 at the New Rifle site are summarized in Table 5–6 are available in Appendix B.

A comparison of river sampling results obtained at on-site and downgradient locations 574 and 548, respectively, indicates the water quality is virtually indistinguishable from background water quality measured up river at locations 545 and 538. Based on these results it can be concluded that contamination associated with the site has had no adverse impact on the river. Any contaminated alluvial ground water that enters the river is quickly diluted to concentrations below detection limits.

Similarly, water quality results from samples collected at the mitigation wetland (1206–1210) are consistent with the range in natural background observed in the reference wetland samples collected upgradient at the One Mile pond (1201–1205). Therefore, it can be concluded that the east end of the mitigation wetland is not impacted by site-related contamination. Surface water at the west end of the mitigation wetland was not present at the time the wetland was sampled in February 1998 and January 1999. Recent reconstruction at the west end of the wetland in May 1999 established a permanent pond and an intermittent oxbow. Surface water quality from these features is addressed as part of the ecological risk assessment in Section 6.0.

Uranium, molybdenum, and all the major elements listed in Table 5–6 are elevated above the upper range in natural background for the surface water samples collected at the Roaring Fork holding pond (1211–1215 and 575). Uranium, molybdenum, and nitrate exceed their respective maximum concentration limits of 0.044, 0.10, and 44 mg/L established for the UMTRA Project (Table 2–1) suggesting that discharge from the contaminated alluvial ground water is impacting the Roaring Fork pond water. However these constituents, which have been measured historically at the Roaring Fork gravel pond since 1991, appear to be decreasing with time as illustrated in Figures 5–20, 5–21, and 5–22. For example, the maximum uranium concentration of 0.435 mg/L observed in October 1991 has decreased to 0.148 mg/L in the most recent water sample collected in January 1999 (Figure 5–20). This observed decrease demonstrates that the alluvial aquifer is naturally flushing clean and eventually will reduce the uranium concentrations to background levels.

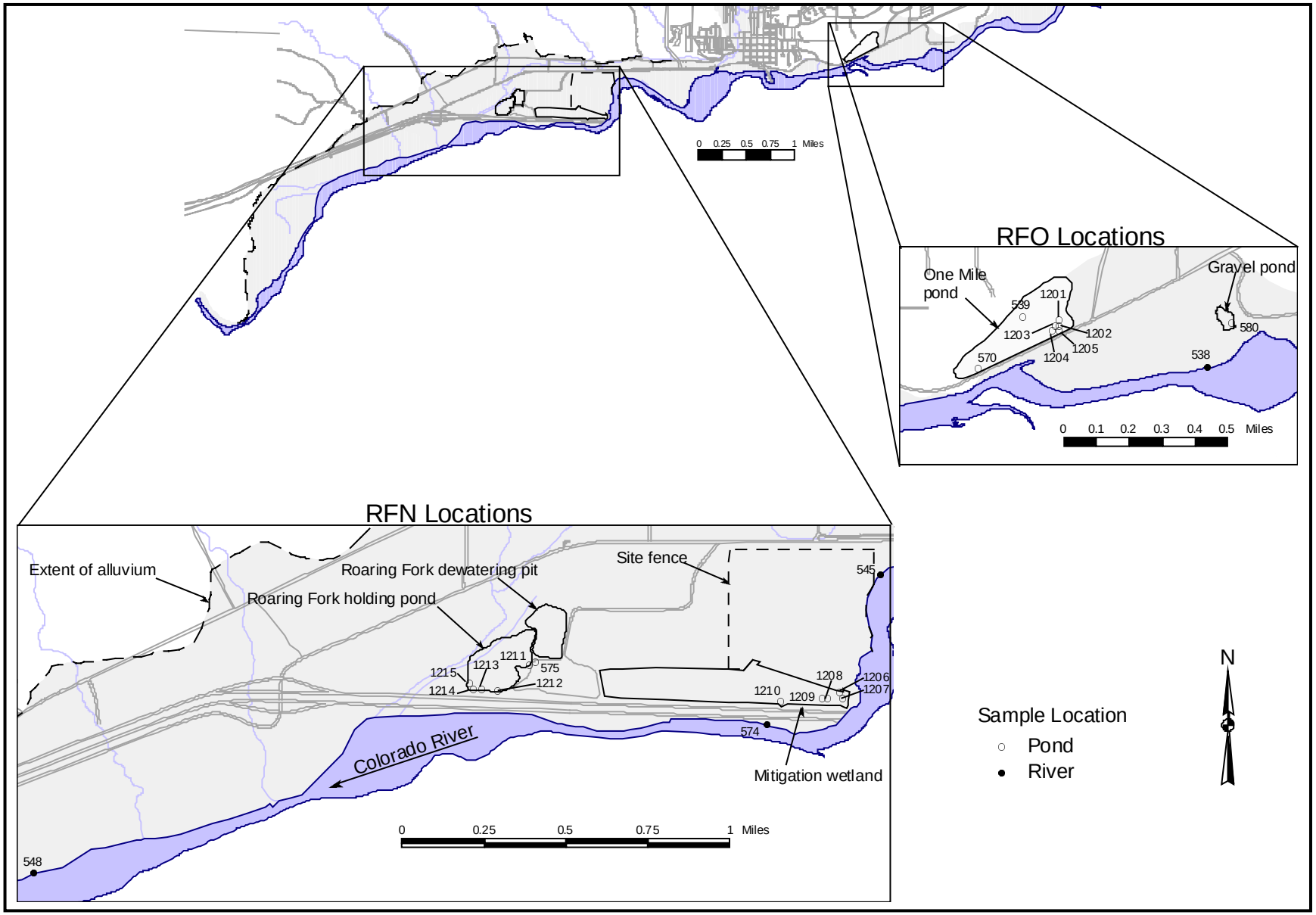


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Figure 5–19. Surface Water Sample Locations

Table 5–6. Surface Water Quality at the New Rifle Site—February 1998 through January 1999

		Site River			Mitigation Wetland			Roaring Fork Gravel Pond[c]			Downgradient River		
Analyte[a]	Unit	FOD	Mean[b]	Range	FOD	Mean[b]	Range	FOD	Mean[b]	Range	FOD	Mean[b]	Range
Major													
Ammonia as NH_4	mg/L	2/2	.04165	.0393-.044	5/5	.1859	.0408-.623	8/8	78.7	33.2-99.5	2/3	.04993	<.0326-.0961
Calcium	mg/L	2/2	65.9	64.2-67.6				3/3	378	363-400	3/3	65.5	63.5-67
Chloride	mg/L	2/2	169	140-198				3/3	302.3	277-334	3/3	157.3	139-178
Magnesium	mg/L	2/2	14.15	13-15.3				3/3	112.7	111-115	3/3	14.53	13.4-15.3
Nitrate	mg/L	1/2	.4656	<.0543-.904	2/5		<.103-2.57	8/8	432.3	380-462	2/3	.5253	<.0578-.973
Sodium	mg/L	2/2	110.5	91.9-129				3/3	944.3	902-989	3/3	109.3	95-127
Sulfate	mg/L	2/2	116.5	106-127	5/5	130.7	91.7-158	8/8	2720	2640-2840	3/3	112.7	105-119
Metal													
Antimony	mg/L	0/2		<.001-<.001				0/3		<.001-<.001	0/3		<.001-<.001
Arsenic	mg/L	0/2		<.0001-<.001	3/5	.006	<.0063-.0086	5/8	.002375	<.0001-.0058	0/3		<.0001-<.001
Cadmium	mg/L	0/2		<.001-<.001	0/5		<.001-<.001	1/8		<.001-.0011	0/3		<.001-<.001
Iron	mg/L	0/2		<.003-<.0185	3/5	.01372	<.011-.0218	1/8		<.003-.013	1/3		.0135-.0173
Lead	mg/L	0/2		<.001-<.001				0/3		<.001-<.001	0/3		<.001-<.001
Manganese	mg/L	2/2	.01095	.0059-.016	5/5	.01322	.007-.0159	8/8	.4676	.159-1.06	3/3	.02187	.0056-.0404
Molybdenum	mg/L	2/2	.008	.0062-.0098	5/5	.05988	.0285-.0775	8/8	.492	.436-.521	3/3	.008267	.0062-.0104
Selenium	mg/L	1/2	.000355	.00021-.001	0/5		<.002-<.002	7/8	.0027	.0015-.004	1/3		.00041-.001
Uranium	mg/L	2/2	.00215	.0018-.0025	5/5	.02738	.0087-.0395	8/8	.1514	.143-.159	3/3	.002133	.0017-.0025
Vanadium	mg/L	0/2		<.001-<.002	2/5		<.005-.011	5/8	.004523	<.001-.0095	0/3		<.001-<.004
Other													
Alkalinity as $CaCO_3$	mg/L	2/2	134	131-137				3/3	289.7	253-318	3/3	147	131-178
Fluoride	mg/L	1/1	.275	.275-.275	5/5	.5626	.423-.674	6/6	.4991	.435-.546			
Redox Potential	mV	2/2	187	142-232				3/3	219.7	183-253	3/3	131.3	33-216
Specific Conductance	umhos/c	2/2	1039	876-1201				3/3	6230	5620-6850	3/3	1030	890-1182
Sulfate/Chloride		2/2	.6993	.6414-.7571				3/3	8.943	8.144-9.531	3/3	.7198	.6685-.7554
Total Dissolved Solids	mg/L	2/2	582.5	503-662				3/3	5040	4790-5440	3/3	566	508-665
pH	s.u.	2/2	8.69	8.45-8.93				3/3	7.817	7.4-8.16	3/3	8.773	8.39-9.21
Radioactive													
Gross Alpha	pCi/L	0/2		<5.48-<6.14				3/3	109.7	49.37-164.6	0/3		<4-<9.56
Gross Beta	pCi/L	0/2		<5.75-<6.02				3/3	88.4	80.09-104.3	1/3		5.18-11.82
Lead-210	pCi/L	0/2		<.91-<1.03				0/3		<.88-<.97	0/3		<.94-<1
Polonium-210	pCi/L	0/2		<.06-<.07				0/3		<.03-<.22	0/3		<.07-<.46
Radium-226	pCi/L	0/2		<.15-<.16				1/3		.09-.14	2/3	.1667	<.1-.24
Radium-228	pCi/L	0/2		<.78-<.85				0/3		<.67-<.8	0/3		<.47-<1
Thorium-230	pCi/L	0/2		<.8-<1.2				0/3		<.8-<1.2	0/3		<.8-<1.2

[a]All measurements are filtered except Specific Conductance, pH, Redox Potential, and Alkalinity as $CaCO_3$ (which are unfiltered).
[b]One-half the detection limit was used to calculate the mean; mean was not calculated if fewer than half of the samples were above detection.
[c]The five downgradient Roaring Fork Pond gravel pond locations (1211–1215) were pooled as one location to calculate the mean.

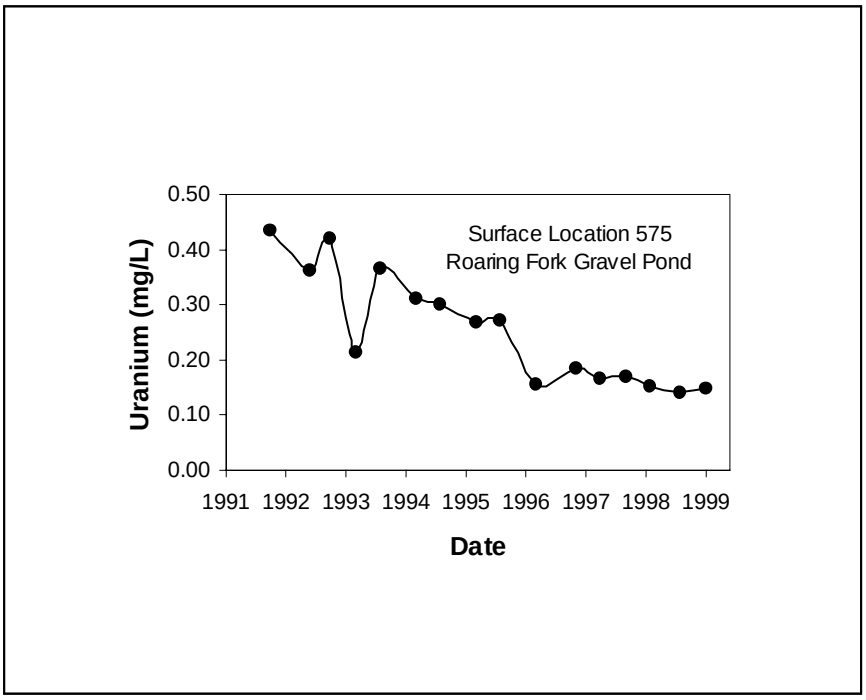

Figure 5–20. Uranium Concentrations Versus Time in the Roaring Fork Gravel Pond Located Downgradient from the New Rifle Site

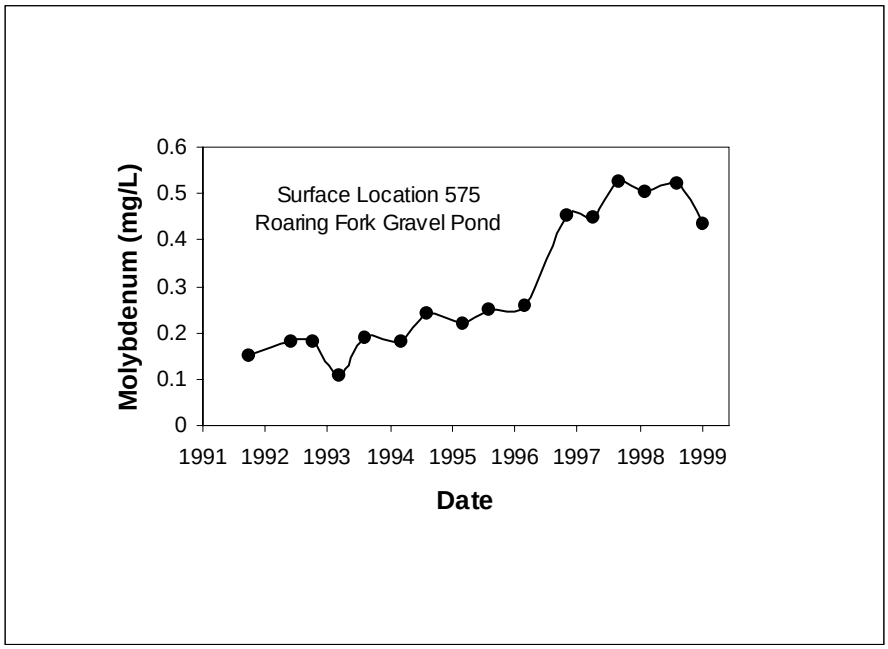

Figure 5–21. Molybdenum Concentrations Versus Time in the Roaring Fork Gravel Pond Located Downgradient from the New Rifle Site

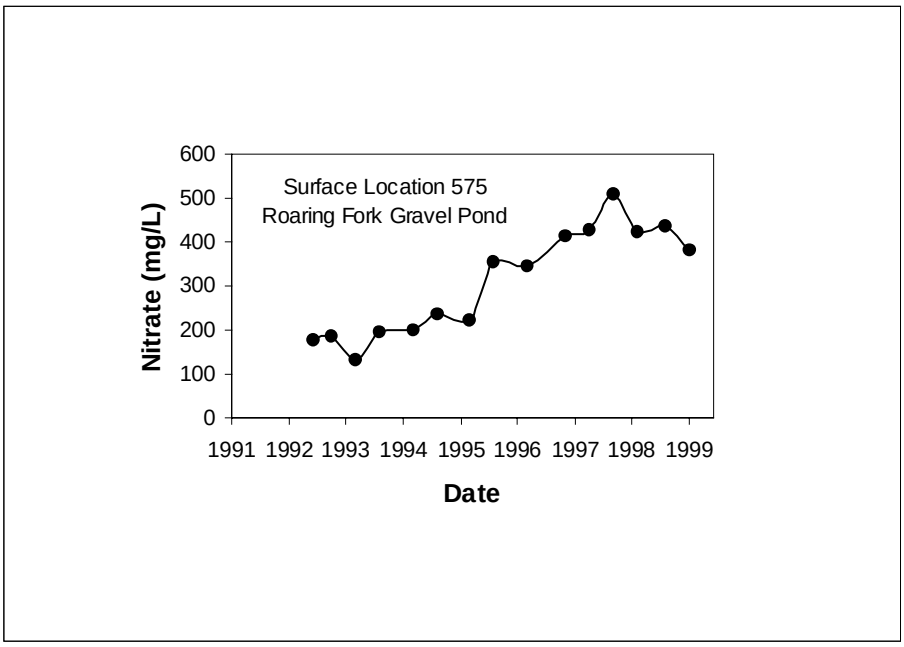

Figure 5–22. Nitrate Concentrations Versus Time in the Roaring Fork Gravel Pond Located Downgradient from the New Rifle Site

Molybdenum and nitrate concentrations in the Roaring Fork gravel pond also appear to be decreasing with time as shown in Figures 5–21 and 5–22. However, concentrations for these contaminants do not start to decrease until after the September 1997 sampling event. A 21 percent decrease in molybdenum concentration, from 0.526 mg/L in September 1997 to 0.436 mg/L in January 1999, is evident in Figure 5–21. A slightly greater decrease of 35 percent in nitrate concentrations, from 511 mg/L to 380 mg/L, is observed over the same 1.4 year time period (Figure 5–22).

5.3.3 Alluvial Aquifer Water Quality

Ground water quality data for the alluvial aquifer in the vicinity of the New Rifle site have been collected since 1985, though frequency and location of wells sampled have varied. The most recent sampling was conducted in May 1998 during the time when the river was near high stage, in August 1998 during average river stage, and in January 1999 during low river stage. Most of the following discussion focuses on results from these three most recent sampling events. Historical data are referenced where appropriate.

5.3.3.1 Background Alluvial Ground Water Quality

Regional background water quality of the alluvial aquifer in the Rifle area is naturally variable and generally poor (Wright Water Engineers 1979). A broad evaluation of the regional ground water in the Rifle area indicates that concentrations of several chemical constituents, including arsenic, barium, molybdenum, radium, selenium, and uranium have exceeded UMTRA standards (DOE 1996d).

Local water quality underneath the site prior to the milling operations is inferred by characterizing the water quality in areas upgradient of the site that are unaffected by process contamination, but which are located in the same flow system that influences the New Rifle site.

Previous work has shown that alluvial water quality from the south side of the Colorado River is significantly different and of substantially higher quality than that on the north side (DOE 1996d). This is attributed to the fact that ground water south of the river receives water from the recharge area that drains basalt mesas; the basalts are relatively insoluble and contribute little to dissolved salts in ground water, unlike the lithologies found on the north side of the river. Therefore, only wells located north of the river are used to characterize background water quality for the site.

Wells RFO–597, –598, –605, and –606 were previously determined to be representative of background (DOE 1996d); since that time well RFO–605 has been removed and additional background wells installed (RFO–291, –292, –658, –659, –660, and RFN–173 and –169). Locations of the background alluvial monitor wells are shown in Figure 5–23. Background alluvial water quality results for the May 1998, August 1998, and January 1999 sampling events are presented in Appendix C and summarized in Table 5–7. These results can be used as a basis for comparison against potential ground water contamination observed at the site.

Sampling results confirm that alluvial ground water in the Rifle area is generally poor. Maximum detected concentrations of uranium and selenium are higher than the UMTRA standards of 0.044 and 0.01 mg/L, respectively. Sulfate values from all background samples greatly exceed the secondary drinking water standard of 250 mg/L. Concentrations for some samples are also higher than the secondary standards for chloride (250 mg/L), iron (0.3 mg/L), and manganese (0.05 mg/L). While the secondary standards are not enforceable or health-based levels, the fact that levels are exceeded provides evidence of the poor quality of the ground water. Most of the other trace elements including antimony, arsenic, cadmium, lead, and vanadium were detected at low concentrations or below the analytical detection limit.

Time-concentration plots for selected constituents are shown in Figures 5–24 through 5–26 for alluvial ground water collected at four background wells having the greatest amount of historical data. These figures show considerable variability in constituent concentration over time, but with no consistent trends indicated.

Water quality results presented in the Piper diagram (Figure 5–27) indicate the dominant anionic species in the alluvial background water is sulfate. This water has a mixed sodium-calcium-magnesium cation composition with a tendency for sodium to dominate. The background alkalinity, as calcium carbonate, ranges from 336 to 645 mg/L. Sulfate concentrations range from 482 to 2,240 mg/L. Calcium concentrations range from 44 to 295 mg/L. Magnesium ranges from 51 to 174 mg/L. On average, the alluvial water pH is near neutral (7.1 s.u.) and the redox condition is slightly reducing (oxidation-reduction potential –48 mV).

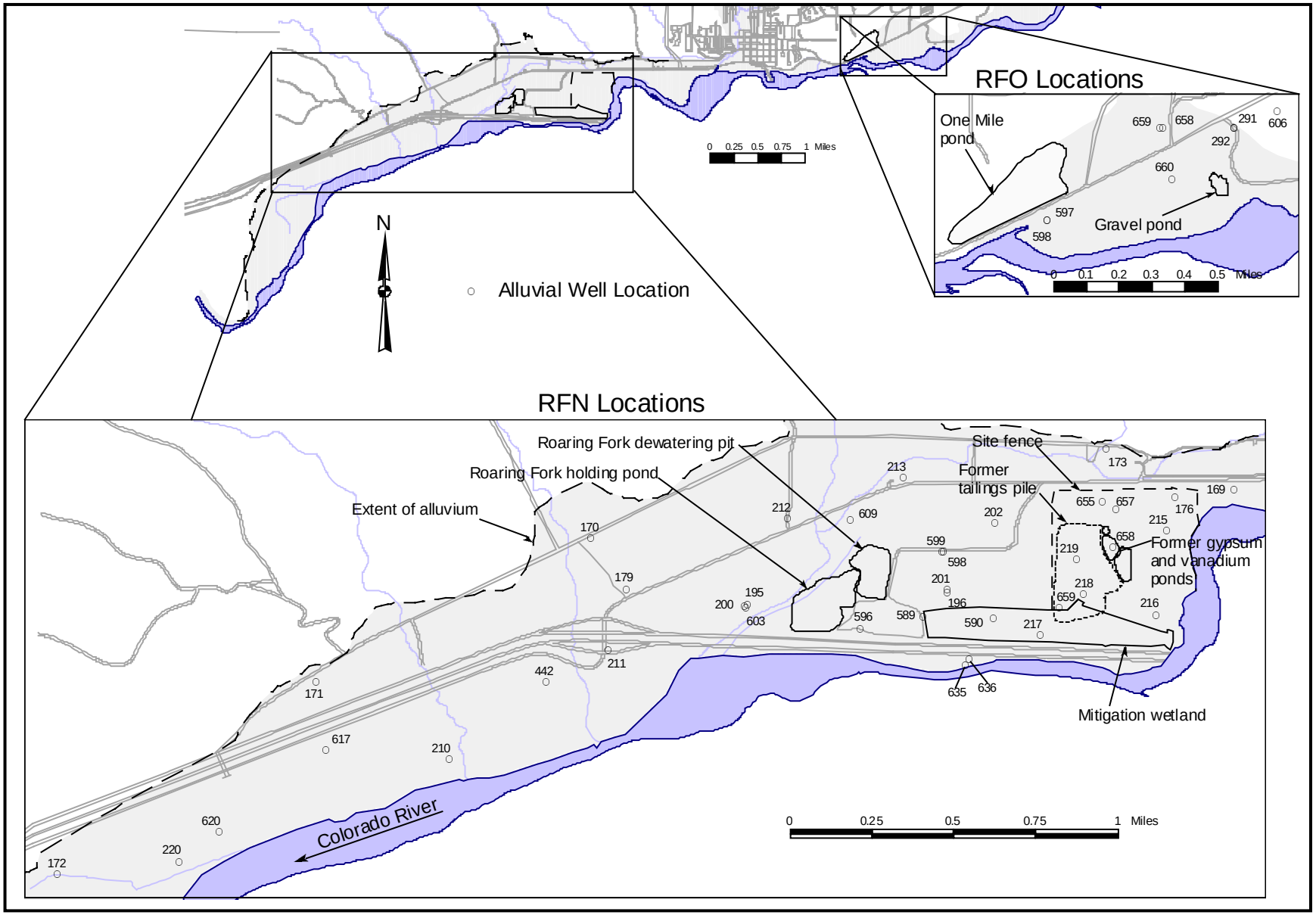

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Figure 5–23. Alluvial Ground Water Monitor Well Sampling Locations

Table 5–7. Background Water Quality for the Alluvial Aquifer—May 1998 through January 1999

Analyte[a]	Unit	Alluvial Background		
		FOD	Mean[b]	Range
Major				
Ammonia as NH_4	mg/L	16/19	.1358	<.001-.61
Calcium	mg/L	19/19	154.7	44.1-295
Chloride	mg/L	19/19	158.8	66-274
Magnesium	mg/L	19/19	122.1	50.6-174
Nitrate	mg/L	15/19	4.025	<.011-20.6
Sodium	mg/L	19/19	424.9	164-655
Sulfate	mg/L	19/19	1101	482-2240
Metal				
Antimony	mg/L	0/19		<.001-<.001
Arsenic	mg/L	8/19		.00031-.0034
Cadmium	mg/L	0/19		<.001-<.001
Iron	mg/L	11/19	.3784	<.003-3.03
Lead	mg/L	1/19		<.001-.0012
Manganese	mg/L	19/19	.8599	.0064-3.66
Molybdenum	mg/L	19/19	.01893	.0034-.0345
Selenium	mg/L	13/19	.01092	<.001-.0298
Uranium	mg/L	19/19	.03719	.0055-.0594
Vanadium	mg/L	5/19		<.001-.0023
Other				
Alkalinity as $CaCO_3$	mg/L	19/19	506.5	336-645
Fluoride	mg/L	19/19	.8409	.422-1.45
Redox Potential	mV	19/19	-47.5	-293-146
Specific Conductance	umhos/cm	19/19	2864	1761-4490
Sulfate/Chloride	s.u.	19/19	8.394	2.82-18.36
pH		19/19	7.101	6.73-8.5
Radioactive				
Gross Alpha	pCi/L	16/19	34.63	<14.86-73.62
Gross Beta	pCi/L	8/19		<12.84-42.69
Lead-210	pCi/L	0/19		<.9-<1.33
Polonium-210	pCi/L	1/19		<.05-.18
Radium-226	pCi/L	7/19		<.04-.17
Radium-228	pCi/L	1/19		<.4-<.8
Thorium-230	pCi/L	0/19		<.8-<2.3
Uranium-234	pCi/L	2/2	21.4	17.9-24.9
Uranium-235	pCi/L	2/2	.635	.52-.75
Uranium-238	pCi/L	2/2	12.85	10.7-15

[a]All measurements are filtered except specific conductance, pH, redox potential, and alkalinity as $CaCO_3$ (which are unfiltered).
[b]One-half the detection limit was used to calculate the mean; mean was not calculated if fewer than half of the samples were above detection

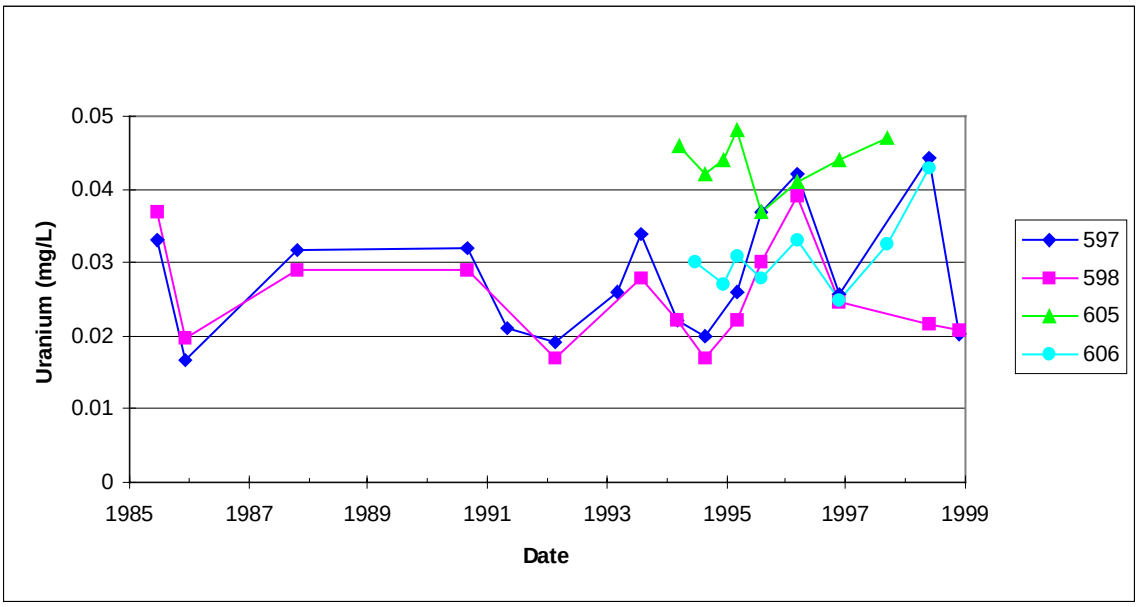

Figure 5–24. Time Concentration Plot for Background Alluvial Ground Water—Uranium

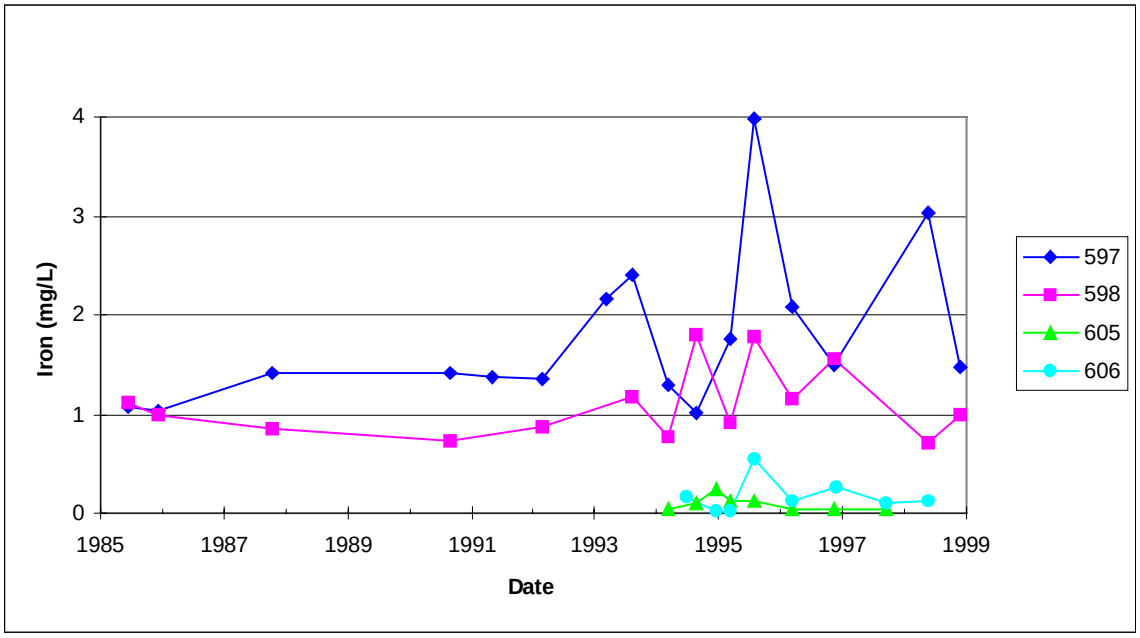

Figure 5–25. Time Concentration Plot for Background Alluvial Ground Water—Iron

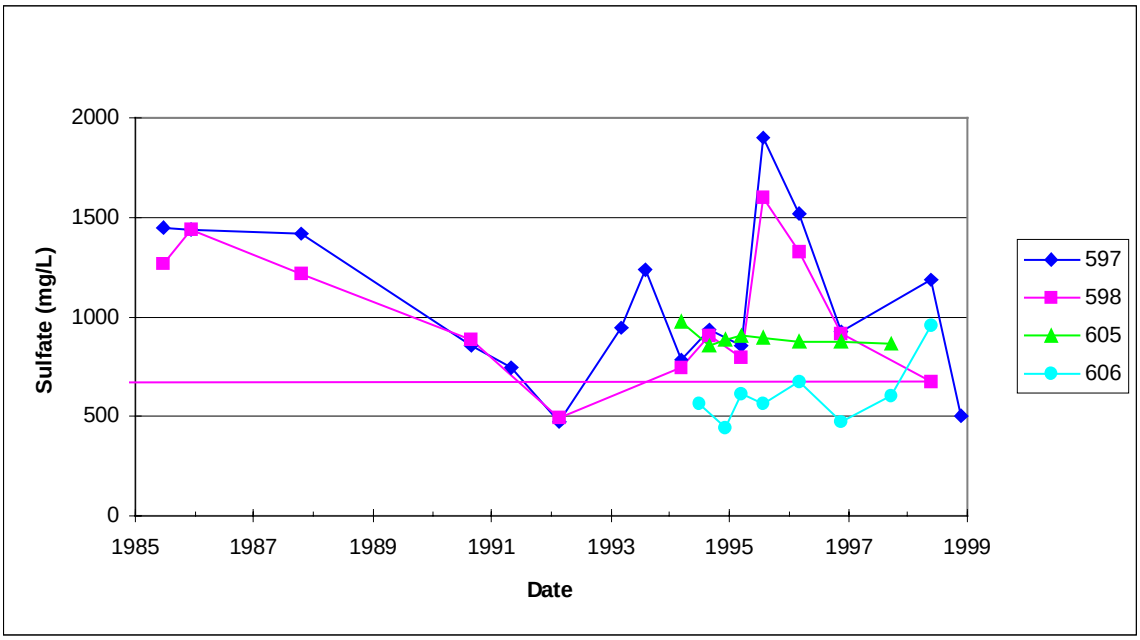

Figure 5–26. Time Concentration Plot for Background Alluvial Ground Water—Sulfate

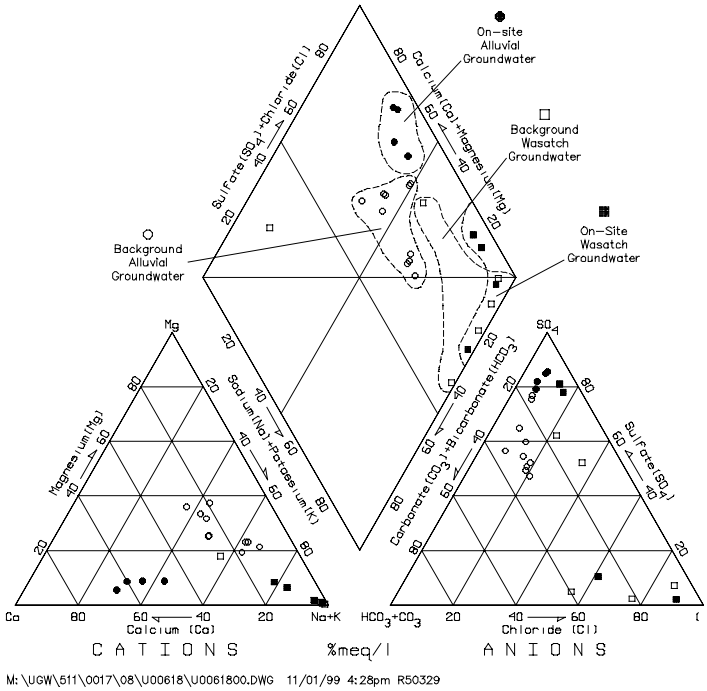

Figure 5–27. Piper Diagram

Geochemical modeling of the alluvial background water from monitor wells RFO–597 and –605, using the PHREEQE software program (Parkhurst 1980), indicates the water is at or near equilibrium with respect to calcite (calcium carbonate), siderite (ferrous iron carbonate), rhodochrosite (manganese carbonate), and gypsum (calcium sulfate) (DOE 1996d). Equilibrium

with these minerals is consistent with the relatively high concentrations of iron, manganese, sulfate, and alkalinity in the background alluvial ground water.

5.3.3.2 Nature and Extent of Alluvial Ground Water Contamination

On-site and downgradient ground water quality has been monitored with varying frequency since 1985. Older wells were monitored until 1992, after which time surface remediation commenced and the on-site wells were abandoned. Surface remediation was completed in 1996. New on-site monitoring wells were installed in 1998 during characterization efforts for this SOWP (Section 4.0). Water from the new wells and from existing downgradient wells installed prior to the 1998 field characterization were sampled in August 1998 and again in January 1999; sampling results are presented in Appendix C and summarized in Table 5–8. Data include results from 37 monitor wells (RFN–170 through–172, –176, –179, –195, –196, –200 through –202, –210 through –213, –215 through –220, –442, –589, –590, –596, –598, –599, –603, –609, –617, –618, –620, –635, –636, –655 and –657 through –659). Monitor well locations are shown in Figure 5–23.

Ammonia, calcium, nitrate, molybdenum, and uranium are the site-related constituents most prevalent in the alluvial aquifer as evidenced by the relatively high frequency (greater than 50 percent) of samples that exceed the upper range in natural background (Table 5–8). The relative low K_ds measured for molybdenum and uranium (Section 4.0) explain the more prevalent occurrence of these site-related constituents as they are conserved in the ground water and more easily dispersed from the former source areas. Similarly, nitrate is geochemically conservative and tends to be highly mobile in ground water under almost all conditions.

Other constituents in the ground water are present at concentrations above the upper range in natural background, however they occur less frequently. For example, sulfate exceeds the upper range in natural background in approximately 49 percent of samples, while sodium exceeds 48 percent, cadmium and fluoride both exceed 34 percent, and chloride and vanadium both exceed 31 percent. All other constituents occur above natural background in less than 30 percent of the samples; manganese exceeds 23 percent, arsenic exceeds 22 percent, magnesium exceeds 18 percent, and selenium exceeds 12 percent. Antimony, iron, and lead all occur above natural background in less than 10 percent of the samples. Several uranium decay products also exceed the upper range in natural background.

Vanadium, ammonia, and molybdenum provide the greatest contrast to natural background. That is, the maximum vanadium concentration of 25.3 mg/L is 11,000 times the upper range in natural background. The maximum ammonia concentration of 669 mg/L is 1,100 times the upper range in natural background and the maximum molybdenum concentration of 6.8 mg/L is 200 times the upper range in natural background. The remaining constituents that exceed natural background exhibit less contrast. For example, the maximum arsenic concentration exceeds the upper range in natural background by 89 times, nitrate exceeds by 35 times, selenium exceeds by 26 times, cadmium exceeds by 18 times, and lead and polonium-210 both exceed by 15 times. Maximum concentrations for calcium, chloride, magnesium, sodium, sulfate, antimony, manganese, uranium, and fluoride exceed the upper range in natural background by less than 10 times.

Table 5–8. On-Site and Downgradient Alluvial Ground Water at New Rifle—August 1998 and January 1999

Analyte[a]	Unit	Alluvial Aquifer				Percent Exceeding Upper Range in Background	Background Range
		FOD	Mean[b]	Range	Max Well		
Major							
Ammonia as NH_4	mg/L	63/65	150.6	<.0038-669	0590	68	<.001-.61
Calcium	mg/L	65/65	311.4	43.5-592	0589	57	44.1-295
Chloride	mg/L	65/65	284.4	27.1-2480	0172	31	66-274
Magnesium	mg/L	65/65	116	12.9-573	0172	18	50.6-174
Nitrate	mg/L	58/65	149.3	<.011-713	0196	57	<.011-20.6
Sodium	mg/L	65/65	734.9	75.1-3120	0172	48	164-655
Sulfate	mg/L	65/65	2264	121-6600	0172	49	482-2240
Metal							
Antimony	mg/L	1/65		<.001-.0011	0442	2	<.001-<.001
Arsenic	mg/L	32/65		<.0001-.304	0658	22	.00031-.0034
Cadmium	mg/L	22/65		<.001-.018	0590	34	<.001-<.001
Iron	mg/L	26/65		<.003-14.3	0655	9	<.003-3.03
Lead	mg/L	3/65		<.001-.018	0442	5	<.001-.0012
Manganese	mg/L	65/65	2.198	.0042-8.82	0590	23	.0064-3.66
Molybdenum	mg/L	62/65	1.026	<.001-6.84	0658	66	.0034-.0345
Selenium	mg/L	44/65	.02744	<.0001-.782	0658	12	<.001-.0298
Uranium	mg/L	65/65	.09804	.0019-.395	0655	72	.0055-.0594
Vanadium	mg/L	22/65		<.001-25.3	0658	31	<.001-.0023
Other							
Alkalinity as $CaCO_3$	mg/L	63/63	499.8	130-1636	0655	11	336-645
Fluoride	mg/L	65/65	1.711	.132-6.39	0636	34	.422-1.45
Redox Potential	mV	65/65	74.25	-280-248	0617	35	-293-146
Specific Conductance	umhos/cm	65/65	4851	29-14860	0172	54	1761-4490
Sulfate/Chloride		65/65	10.18	.7391-21.48	0658	5	2.82-18.36
pH	s.u.	65/65	6.964	6.04-7.81	0215	0	6.73-8.5
Radioactive							
Gross Alpha	pCi/L	52/64	60.18	<10-229.9	0655	19	<14.86-73.62
Gross Alpha – Uranium	pCi/L	52/65		0-91.79	0655		
Gross Beta	pCi/L	41/64	48.55	<5.8-141.3	0655	42	<12.84-42.69
Lead-210	pCi/L	5/64		<.86-1.24	0609	0	<.9-<1.33
Polonium-210	pCi/L	5/64		<.04-2.75	0658	3	<.05-.18
Radium-226	pCi/L	4/64		<.08-<.17	0620	2	<.04-.17
Radium-228	pCi/L	24/64		<.39-3.64	0659	19	<.4-<.8
Thorium-230	pCi/L	2/65		<.8-<2	0655	0	<.8-<2.3
Uranium-234	pCi/L	6/6	17.78	1.6-41.1	0172	33	17.9-24.9
Uranium-235	pCi/L	3/6	.7817	<.4-1.6	0201	50	.52-.75
Uranium-238	pCi/L	6/6	15.35	1.3-32.7	0201	50	10.7-15

[a]All measurements are filtered except specific conductance, pH, redox potential, and alkalinity as $CaCO_3$ (which are unfiltered).
[b]**One-half the detection limit was used to calculate the mean; mean was not calculated if fewer than half of the samples were above detection**

Ground water concentrations of arsenic, cadmium, lead, molybdenum, nitrate, selenium, uranium, gross alpha, and combined radium-226 and radium-228 are regulated by UMTRA standards (Table 2–1). Nitrate, arsenic, cadmium, molybdenum, selenium, uranium, and gross alpha activity are site-related constituents that exceed their respective standards of 44 mg/L, 0.05 mg/L, 0.01 mg/L, 0.1 mg/L, 0.01 mg/L, 0.044 mg/L, and 15 pCi/L. Lead occurs above natural background, but the maximum concentration of 0.018 mg/L is relatively low and less than the UMTRA ground water standard of 0.05 mg/L. The remaining regulated constituents,

combined radium-226 and radium-228, are present in the alluvial ground water at relatively low concentrations below the UMTRA ground water standard of 5 pCi/L, or were not detected.

Ammonia, arsenic, molybdenum, nitrate, selenium, uranium, and vanadium are selected as the focus for the remainder of the discussion on contamination in the alluvial aquifer. Arsenic, molybdenum, nitrate, selenium, and uranium are selected because they are site-related constituents that exceed their respective UMTRA ground water standards. Ammonia and vanadium are selected because they are present at concentrations significantly greater than natural background.

Ammonia

Distribution of ammonia concentrations in the alluvial aquifer is shown in Figure 5–28 for the August 1998 sampling event. The colored areas on the figure exceed the approximately 200 mg/L risk-based concentration (RBC) for human health (Section 6.1). The highest concentrations in the plume appear to originate near the center of the former gypsum-vanadium evaporation pond and extend in a southwesterly direction through the western portion of the mitigation wetland. The highest ammonia concentration of 669 mg/L is located at monitor well RFN–590, which is centered near the west end of the mitigation wetland, approximately 2,100 ft downgradient from the former source area. Elevated concentrations extend west of the mitigation wetland boundary to the Roaring Fork gravel operation where the concentrations decrease to less than 50 mg/L.

A similar ammonia distribution pattern is observed for January 1999. However, the concentrations in the plume tend on average to be slightly lower in 1999 than in 1998. For example, the highest concentration at monitor well RFN–590 decreased to 627 mg/L in 1999 from 669 mg/L in 1998. Similar decreases in ammonia concentrations are evident in 1999 at most of the other locations in the plume with two notable exceptions. Ammonia concentrations in ground water sampled from monitor well RFN–658, which is located near the center of the former gypsum-vanadium evaporation pond, increased to 367 mg/L in 1999 from 283 mg/L in 1998. This suggests evidence for a more recent plume that may have been mobilized when the surface remediation was in progress. Perhaps water-soluble constituents in the tailings were mobilized by construction water applied during the surface remediation and by irrigation during the reseeding operation at the conclusion of the surface remediation. The other exception is an increase observed in 1999 at monitor well RFN–635, which is located along the river approximately 800 ft downgradient from the mitigation wetland. Ammonia concentration at this most downgradient location increased to 550 mg/L in 1999 from 502 mg/L in 1998, suggesting the center of the plume is naturally flushing to the river.

Arsenic

Arsenic concentrations (Figure 5–29) in ground water at most wells are at the detection limit (0.001 mg/L) and only two locations (RFN–658 and –659) had concentrations of arsenic in ground water exceeding the UMTRA standard of 0.05 mg/L. The maximum detected concentration of arsenic for the August 1998 sampling event was 0.30 mg/L at well RFN–658 which is located near the former gypsum-vanadium evaporation pond.

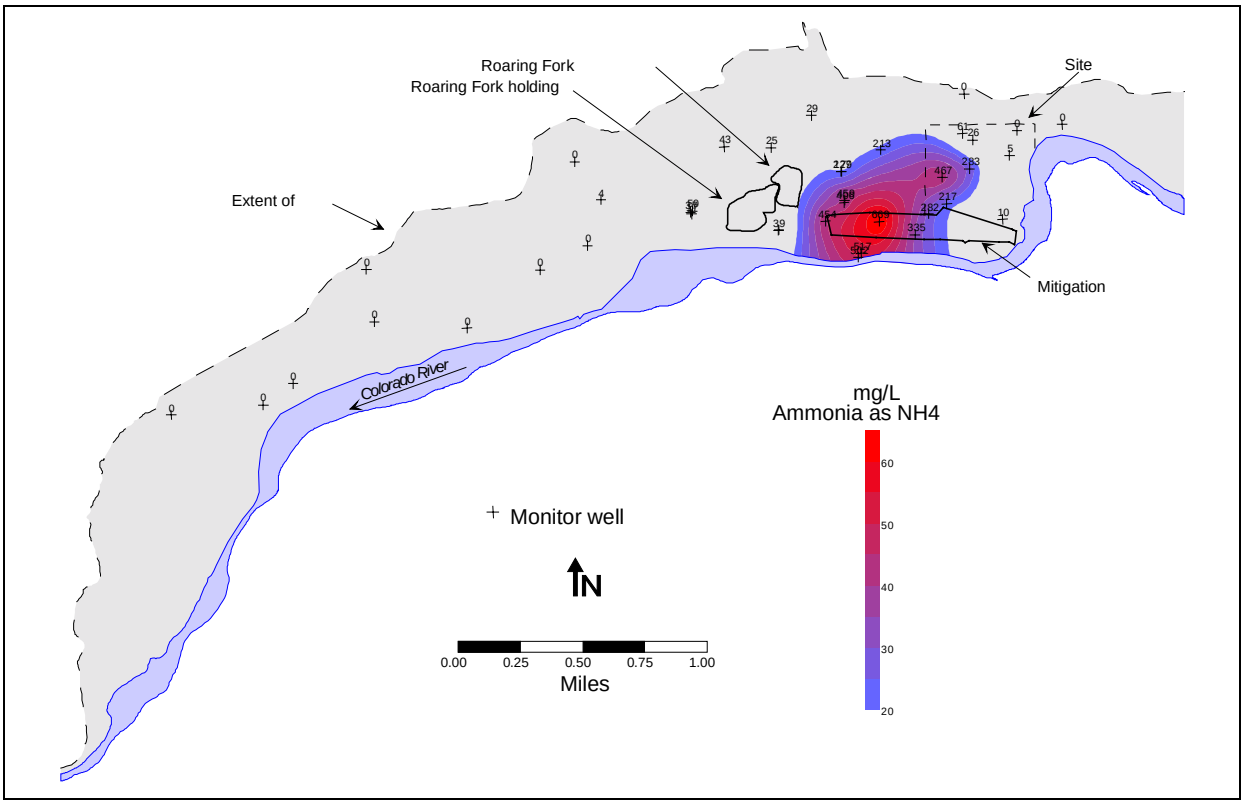

Figure 5–28. Ammonia Distribution in Alluvial Ground Water—New Rifle, 1998

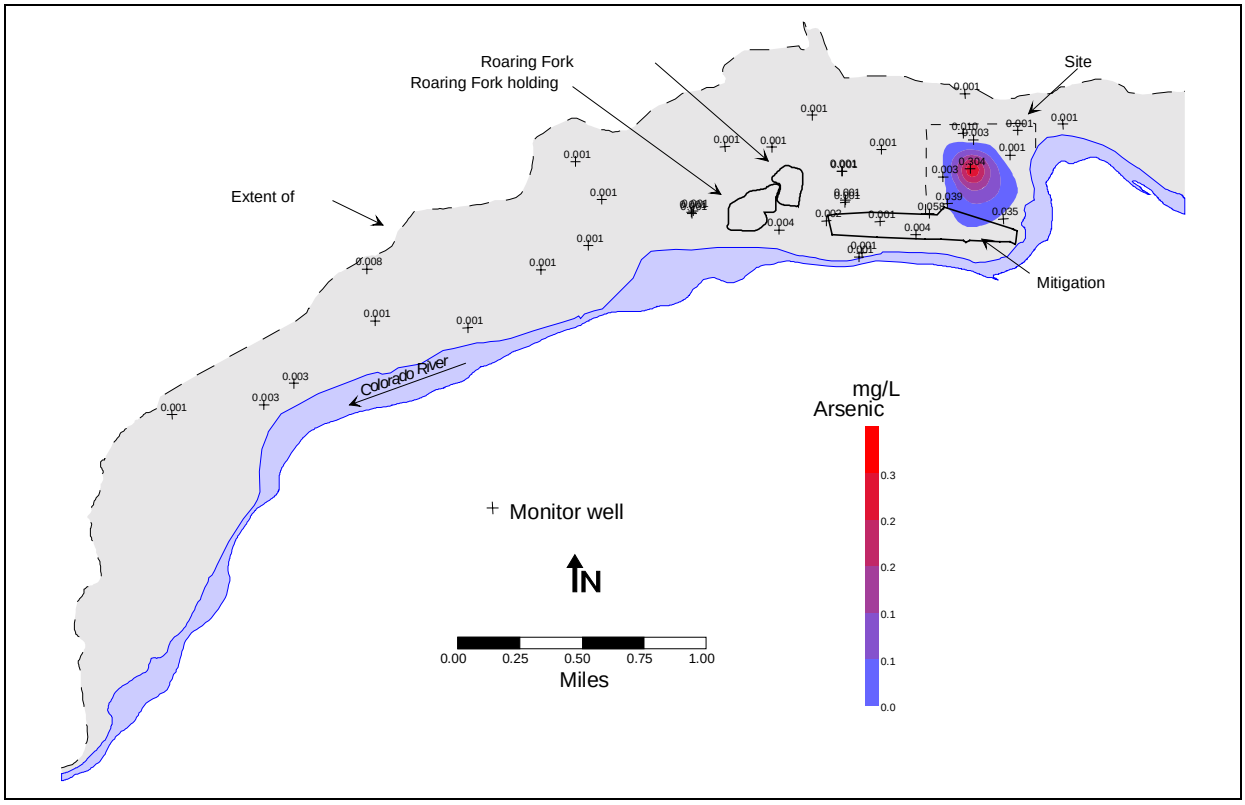

Figure 5–29. Arsenic Distribution in Alluvial Ground Water—New Rifle, 1998

In January 1999 the arsenic concentration at well RFN–658 decreased to 0.08 mg/L, a value only slightly above the UMTRA standard. Arsenic concentrations in ground water from the January 1999 sampling event are presented in Figure 5–30. Elevated arsenic concentrations observed in 1998 and 1999 do not extend beyond the boundary of the New Rifle site. The limited distribution of arsenic may be attributed to slow plume migration and adsorption by the alluvial aquifer matrix, an interpretation that is consistent with the relative immobility of arsenic as predicted by the K_d (Section 4.4).

Molybdenum

Molybdenum concentrations are distributed in the alluvial aquifer as shown in Figure 5–31for the period August 1998. The colored areas in Figure 5–31 delineate molybdenum concentrations above the UMTRA ground water standard of 0.1 mg/L. The highest concentrations ranging between 3.6 and 6.4 mg/L appear to be located beneath the former tailing area (RFN–218, –219, –659) and the former gypsum-vanadium evaporation ponds (RFN–658). The plume extends off site in a southwest-west direction as far downgradient as the Roaring Fork gravel operation. Concentrations up to 0.18 mg/L molybdenum detected in ground water from monitor well RFN–195, which is located just west of the Roaring Fork holding pond, represents the most downgradient leading edge of the plume that exceeds the UMTRA standard. Similar molybdenum concentrations and distribution patterns are observed for the January 1999 ground water sampling results.

Nitrate

Distribution of nitrate concentrations in the alluvial aquifer is shown in Figure 5–32 for the August 1998 ground water sampling event. In addition, results of the field analyses performed during the 1998 field investigation (Table 4–1) are included in Figure 5–32 to obtain the most comprehensive coverage possible. The colored areas on the figure exceed the UMTRA standard of 44 mg/L.

It is evident from Figure 5–32 that most of the nitrate contamination has migrated offsite in a southwest-west direction. The most striking feature in Figure 5–32 is the delineation of two distinct plumes separated by the Roaring Fork gravel ponds. The highest nitrate concentration of 635 mg/L (RFN–589) is located in the east plume in an area between the Roaring Fork dewatering pond and the west end of the mitigation wetland. Ground water flow velocities in this area of the alluvial aquifer increase toward the Roaring Fork dewatering pond due to the active mining operation, especially during higher river stage (Section 5.2.2.1), thereby enhancing the natural flushing of nitrate. Plume water actively drawn into the dewatering pond is then pumped into the downgradient holding pond at which point the water evaporates and infiltrates back into the alluvial aquifer. The ground water mounding effect from the holding pond tends to increase the ground water flow velocities in a radial pattern downgradient from the holding pond, thereby reducing nitrate concentrations through dilution and dispersion. The maximum nitrate concentration associated with the west plume is 377 mg/L at monitor well RFN–195.

Two small anomalous areas where nitrate concentration in the ground water is slightly elevated above the UMTRA standard of 44 mg/L occur over 2 miles downgradient of the New Rifle site. Nitrate concentrations measured in 1998 at these locations are 48 and 48.6 mg/L, at temporary well location RFN–184 and monitor well RFN–617, respectively. Monitor well location RFN–617 was installed in 1998 as a replacement for private well RFN–428 that was damaged in 1997. The most recent nitrate concentrations in ground water collected at this location are higher

than the historical levels of 12 mg/L reported in 1991 and 1992 when the original well was first sampled. However, as illustrated in Figure 5–33, nitrate levels appear to be declining since the highest concentration of 75 mg/L was detected in 1995. It is possible that increased ground water flow velocities downgradient from the Roaring Fork holding pond has contributed to the slightly elevated nitrate concentrations detected at this location. Conversely, the nitrate could originate from sources other than the former milling activities, such as septic tanks, cattle stock ponds, or fertilizer. In any case, the nitrate levels at these two downgradient locations only slightly exceed the UMTRA standard and appear to be limited in areal extent.

Selenium

The distribution of selenium in August 1998 exceeding the 0.05 mg/L EPA SDWA ground water standard (Table 2–1) is delineated by the colored areas in Figure 5–34. A larger area exceeds the UMTRA standard of 0.01 mg/L, including many background locations; therefore, the area exceeding the 0.05 mg/L standard is a better indicator of the extent of site-related contamination. The distribution is somewhat similar to that of arsenic, in that elevated concentrations are generally confined to the site. Concentrations exceeding the 0.05 mg/L standard is exceeded at only three on-site locations. The highest concentration of 0.78 mg/L selenium was detected at monitor well RFN–658 which is located near the center of the former gypsum-vanadium evaporation pond.

In January 1999 the concentration detected at monitor well RFN–658 decreased to 0.16 mg/L. Selenium concentrations from the January 1999 sampling event are presented in Figure 5–35. Similar to arsenic, the limited distribution of selenium may be attributed to slow plume migration and adsorption by the alluvial aquifer matrix, an interpretation that is consistent with the relative immobility of selenium as predicted by the K_d (Section 4.4).

Uranium

The uranium plume associated with the site is shown in Figure 5–36 for August 1998. In addition, results of the field analyses performed during the 1998 field investigation (Table 4–1) are included in Figure 5–36 to obtain the most comprehensive coverage possible. The colored areas on the figure exceed the UMTRA ground water standard of 0.044 mg/L uranium.

It is evident from Figure 5–36 that the uranium dispersion is similar to the nitrate dispersion with respect that contamination has migrated a significant distance off site and the Roaring Fork gravel ponds separate two distinct plumes. The highest uranium concentration of 0.40 mg/L (RFN–655) is located in the east plume near the northern boundary of the former tailing area and extends southwest-west to the Roaring Fork dewatering pond. As with nitrate, natural flushing of uranium is being enhanced by the increased ground water flow velocities created by the dewatering activities at the Roaring Fork gravel pond (Section 5.2.2.1). Similarly, the radial dispersion of uranium exhibited by the west plume around the downgradient holding pond reflects the increase in ground water velocities created by the ground water mound which reduces uranium concentrations through dilution and dispersion. The maximum uranium concentration associated with the west plume is 0.227 mg/L at monitor well RFN–596.

Vanadium

The distribution of vanadium in the alluvial ground water exceeding the risk-based level of 0.33 mg/L based on human health for drinking water is shown by the colored areas in Figure 5–37.

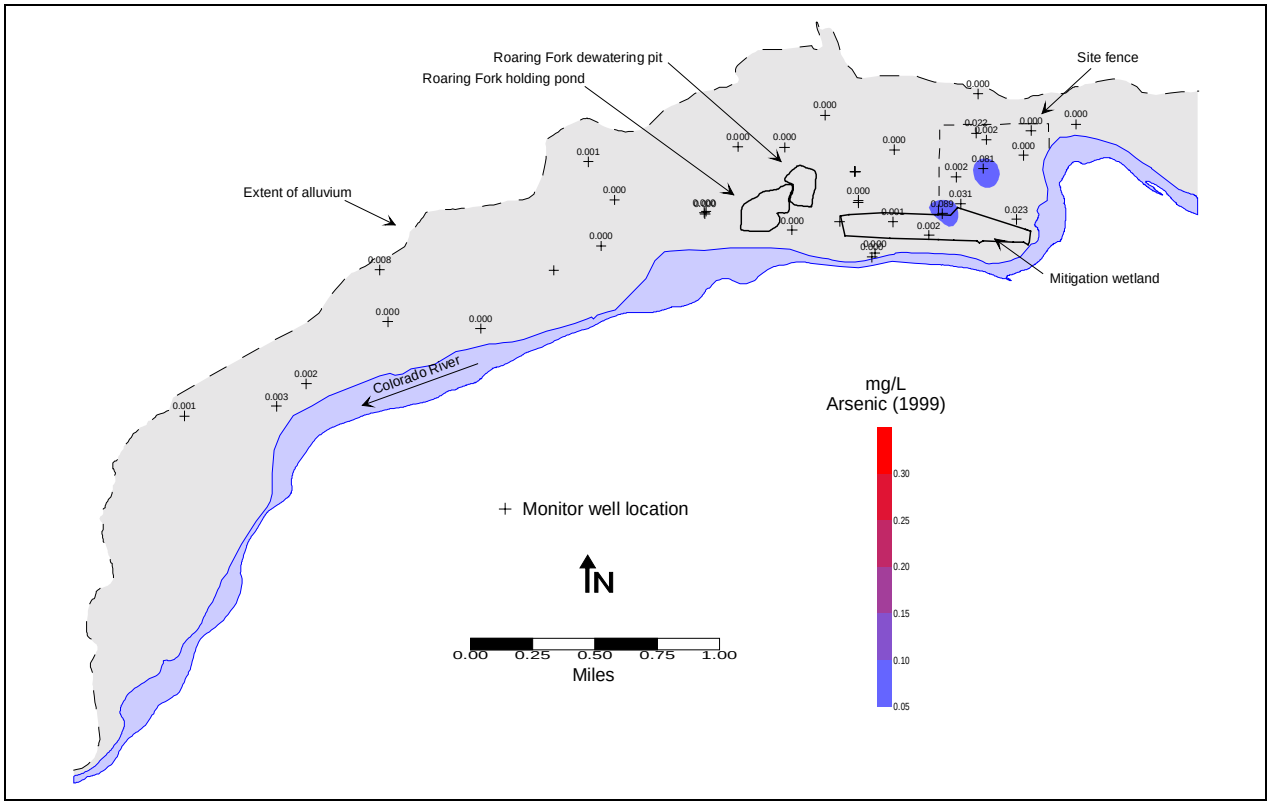

Figure 5–30. Arsenic Distribution in Alluvial Ground Water—New Rifle, 1999

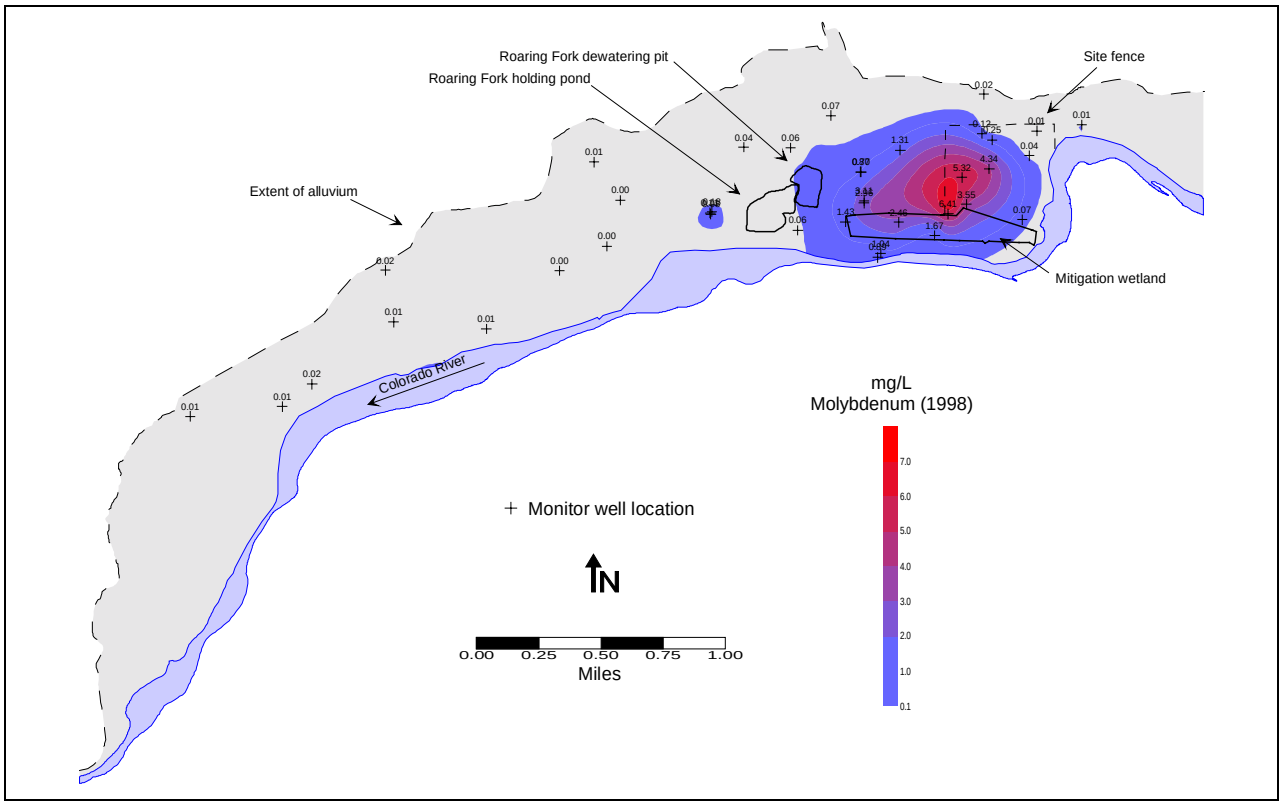

Figure 5–31. Molybdenum Distribution in Alluvial Ground Water—New Rifle, 1998

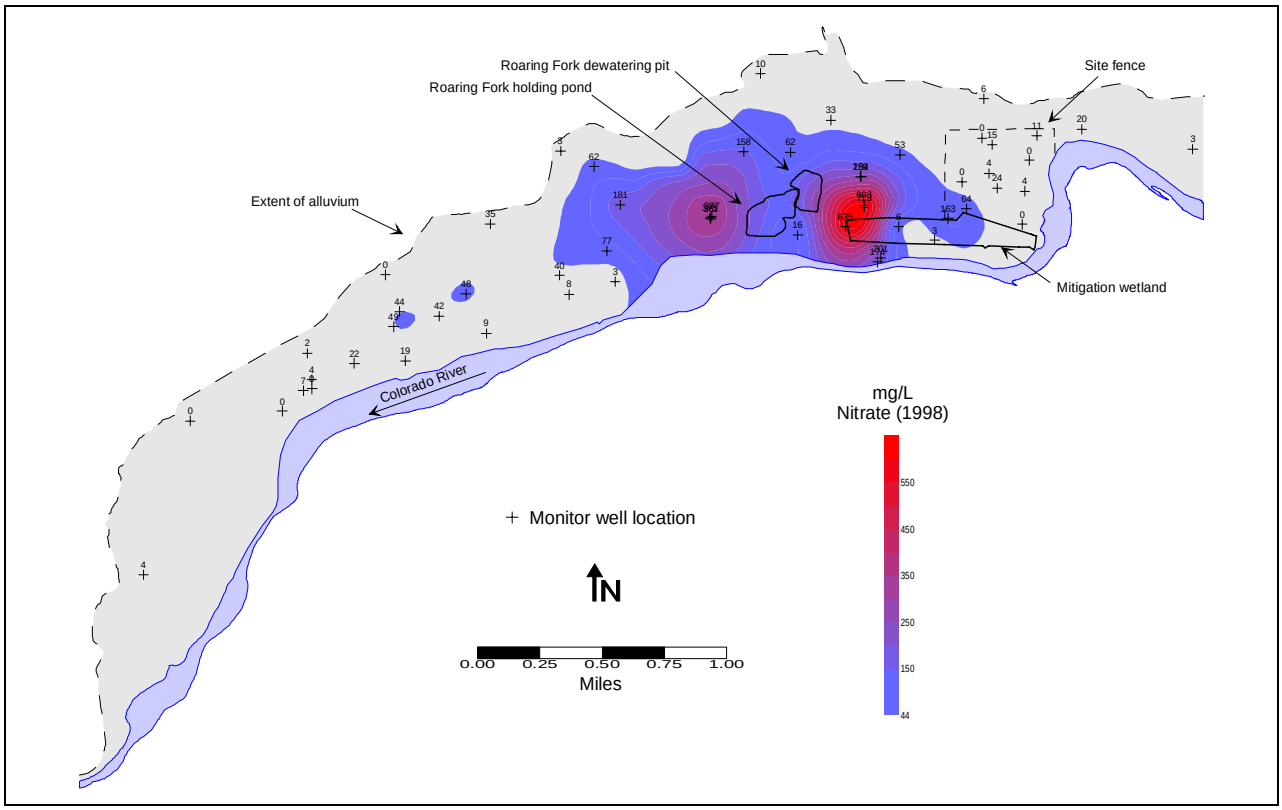

Figure 5–32. Nitrate Distribution in Alluvial Ground Water—New Rifle, 1998

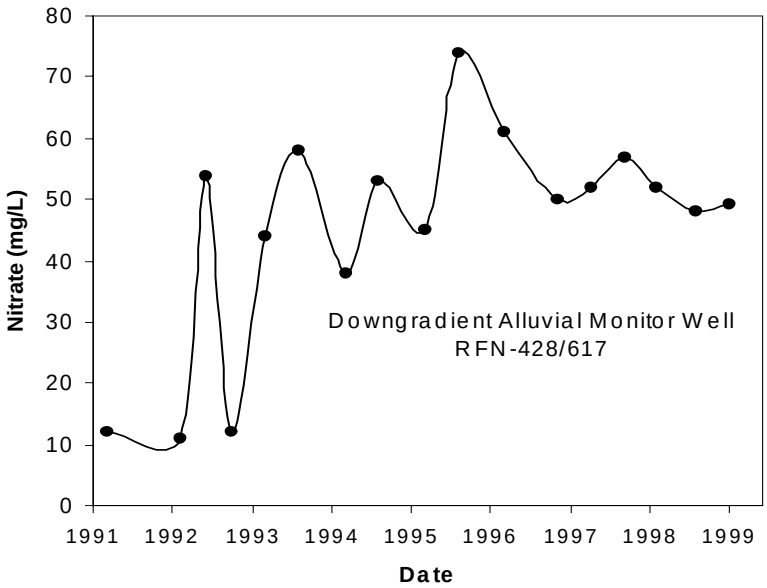

Figure 5–33. Nitrate Concentrations Versus Time at Private Well RFN–428 and –617 Located Approximately 2 Miles Downgradient from the New Rifle Site

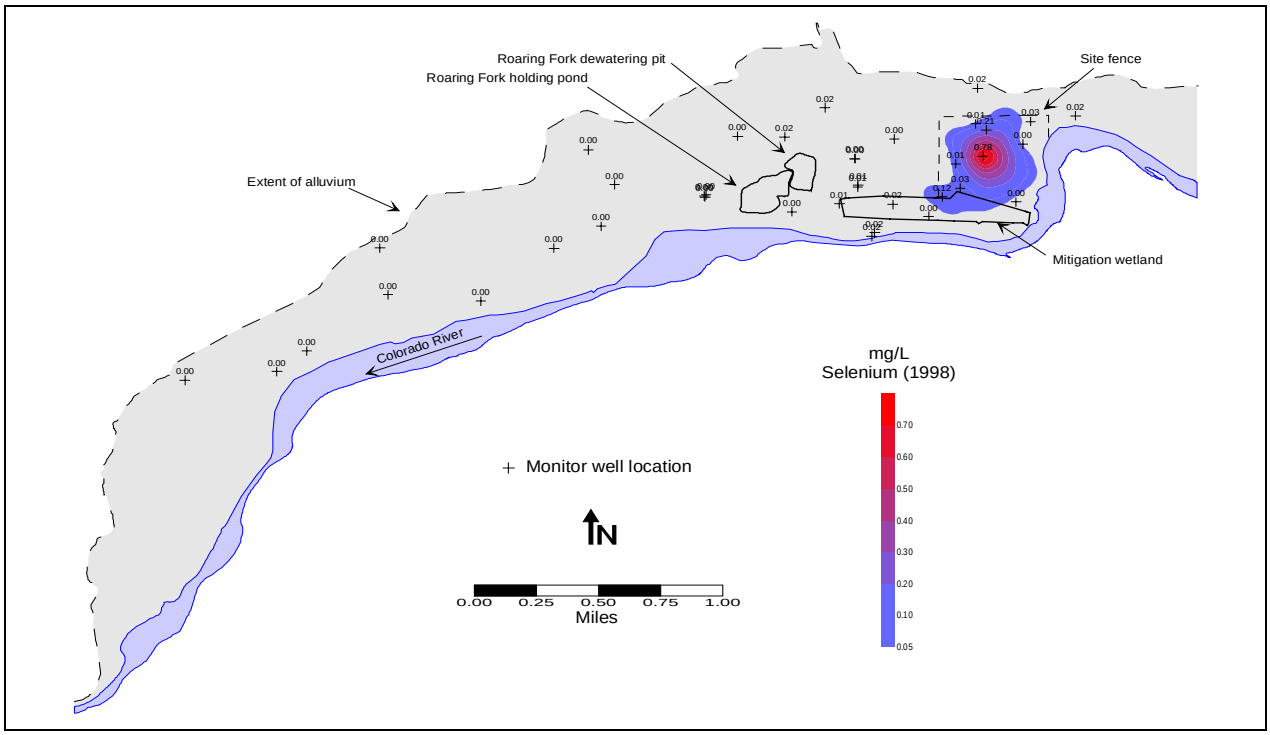

Figure 5–34. Selenium Distribution in Alluvial Ground Water—New Rifle, 1998

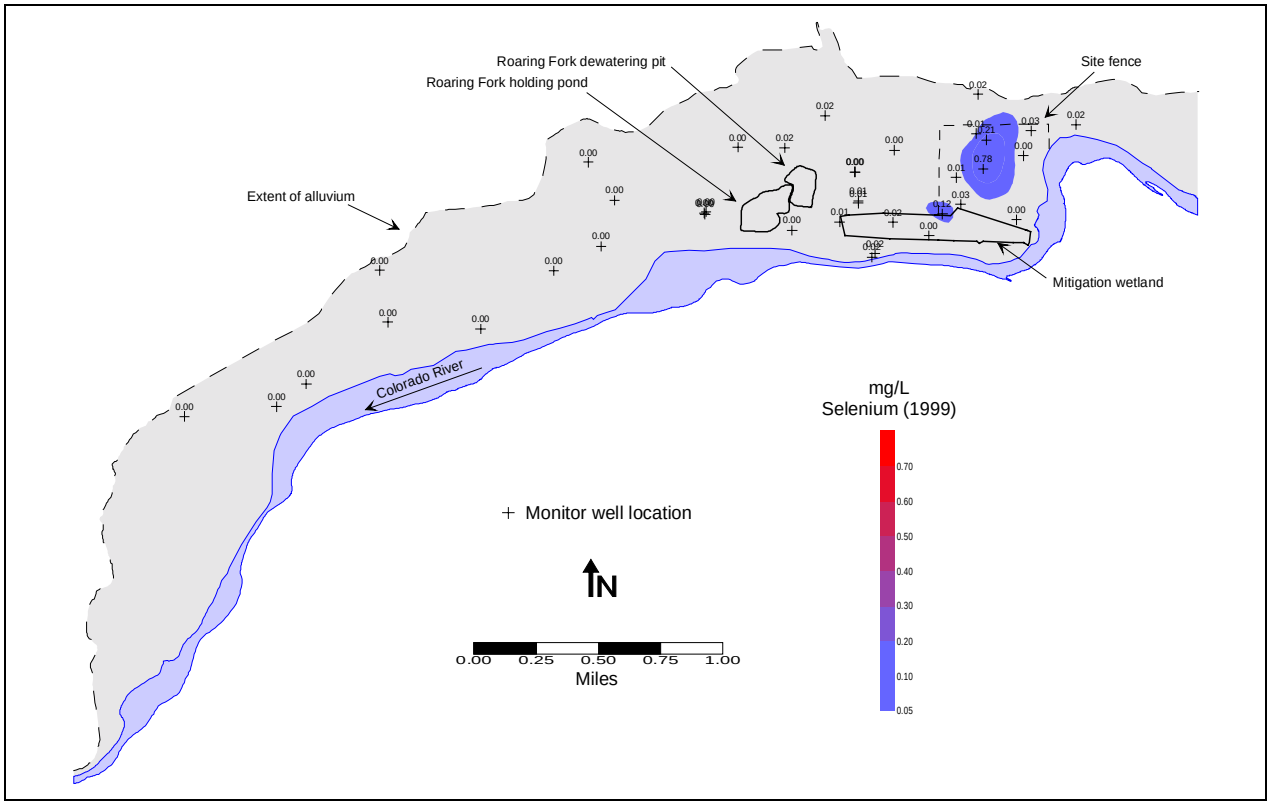

Figure 5–35. Selenium Distribution in Alluvial Ground Water—New Rifle, 1999

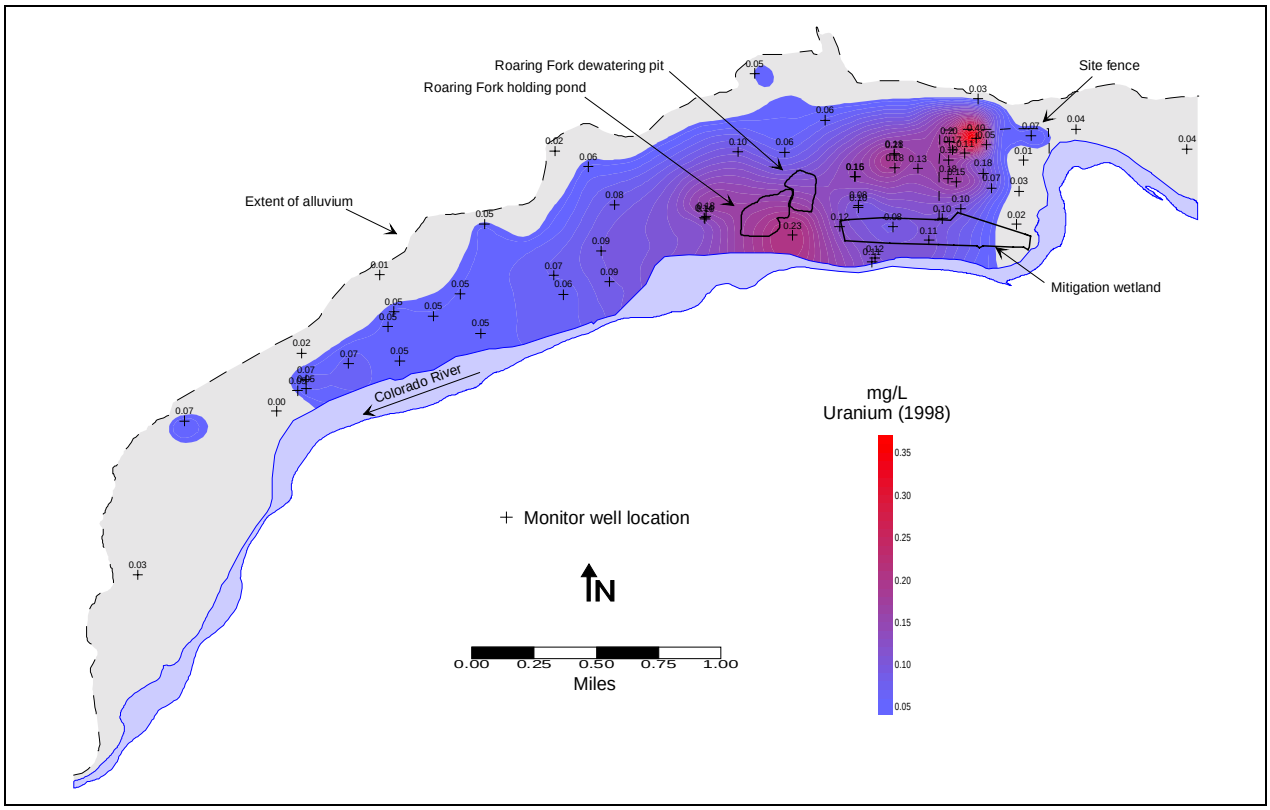

Figure 5–36. Uranium Distribution in Alluvial Ground Water–New Rifle, 1998

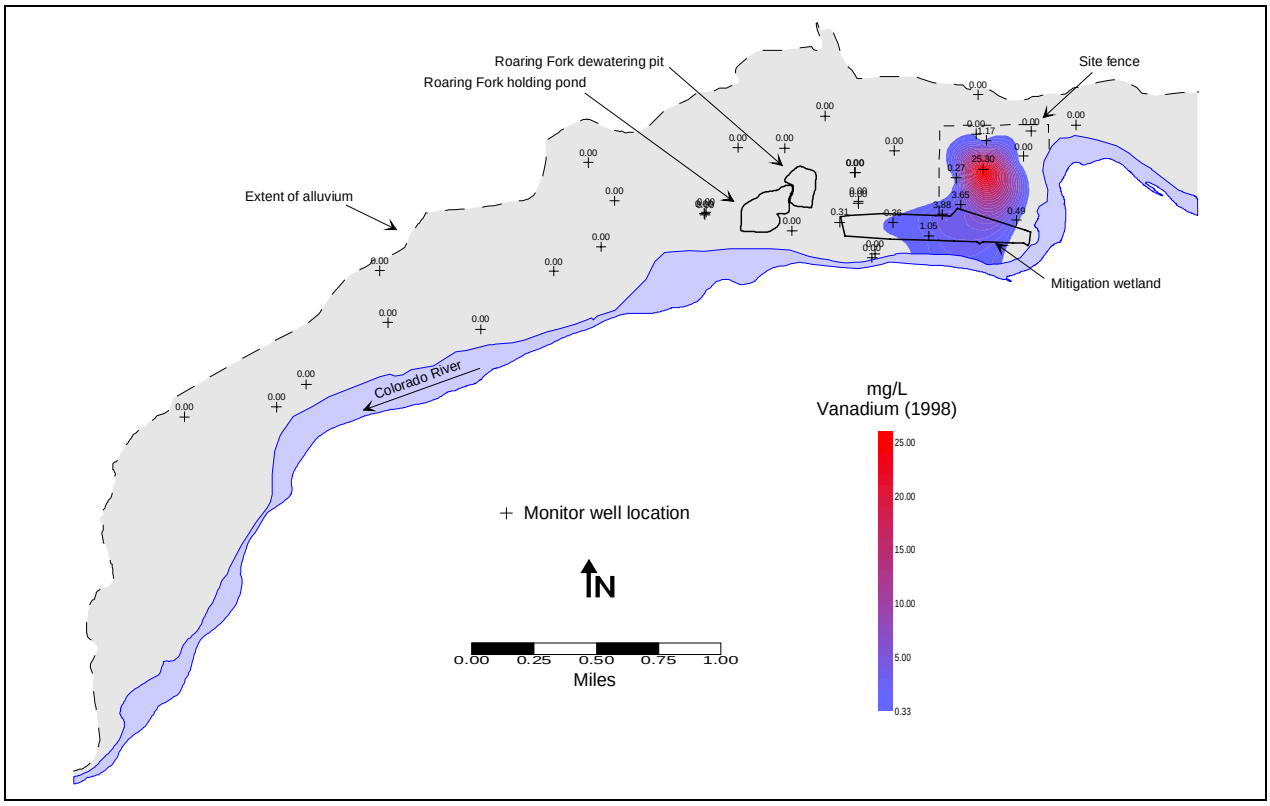

Figure 5–37. Vanadium Distribution in Alluvial Ground Water–New Rifle, 1998

The distribution of the vanadium plume bears some similarity to that of arsenic and selenium in that the elevated concentrations are mostly confined to the site. In the case of vanadium however, an area in the mitigation wetland southwest of the site also exceeds the 0.33 mg/L RBC.

The maximum vanadium concentration of 25.3 mg/L was detected August 1998 in the ground water sample collected from monitor well RFN–658 which is located near the former gypsum-vanadium evaporation pond. This is also the same ground water monitor well where the maximum arsenic and selenium concentrations were observed. It is interesting to note that monitor well RFN–658 was installed in 1996 near historical on-site wells RFN–593 and –594, which were removed during the surface remediation. All three monitor wells were completed with a 5 ft screen interval near the upper portion of the alluvial aquifer and therefore, water quality results from these wells can be considered representative of the same location. RFN–658 however, is a stainless steel well point that was installed using a backhoe. The installation procedure consisted of excavating a pit near the water table then setting the well point vertically in the pit and driving the point approximately 1 ft deeper. The well point and pit was then backfilled with cuttings to the original surface. Approximately 2 ft of the upper saturated thickness is screened at RFN–658.

Vanadium concentration versus time at this location is illustrated in Figure 5–38. The data reveal a continuous decrease in vanadium concentrations in the historical wells. Concentrations decreased from 20 mg/L in 1985 to 0.73 mg/L in 1992 at which time the wells were removed during the surface remediation. Concentrations increased significantly to 44 mg/L vanadium in 1996 when the well point was installed. This significant increase in vanadium concentration suggests evidence for a more recent plume that may have been mobilized when the surface remediation was in progress. Perhaps constituents in the tailings were mobilized by construction water applied during the surface remediation and by irrigation during the reseeding operation at the conclusion of the surface remediation. An alternate explanation is that some of the subpile soil cuttings used to backfill the well point contained minor amounts of residual vanadium thereby providing a small isolated point source. In any case, the vanadium concentrations at this location appear to be again decreasing with time. In January 1999 the vanadium concentration at monitor well RFN–658 decreased to 12.9 mg/L.

5.3.4 Water Quality of the Wasatch Formation

5.3.4.1 Background Water Quality in the Wasatch Formation

Background water quality in the Wasatch Formation can be evaluated by examining analytical results of water samples collected from upgradient monitor wells RFO–620, –621, –622 and RFN–640 and –641. These wells are all completed in the upper Wasatch that is hydraulically connected to the alluvial aquifer, but at different depths. Wasatch monitor wells RFO–629, –630, and –641 are all located south of the Colorado River. As with the alluvial aquifer south of the river, these wells receive water from the recharge area that drains basalt mesas that are geochemically distinct from the drainage systems north of the river. Because the Wasatch Formation near the site is influenced primarily by the drainage systems north of the site, the wells south of the river were excluded from the background analysis. Locations of the background monitor wells used in the evaluation are shown in Figure 5–39. Available water quality results for samples collected May 1998, August 1998, and January 1999 at monitor wells RFO–620, –621, –622 and RFN–640 and –641 are presented in Appendix C and summarized in Table 5–9.

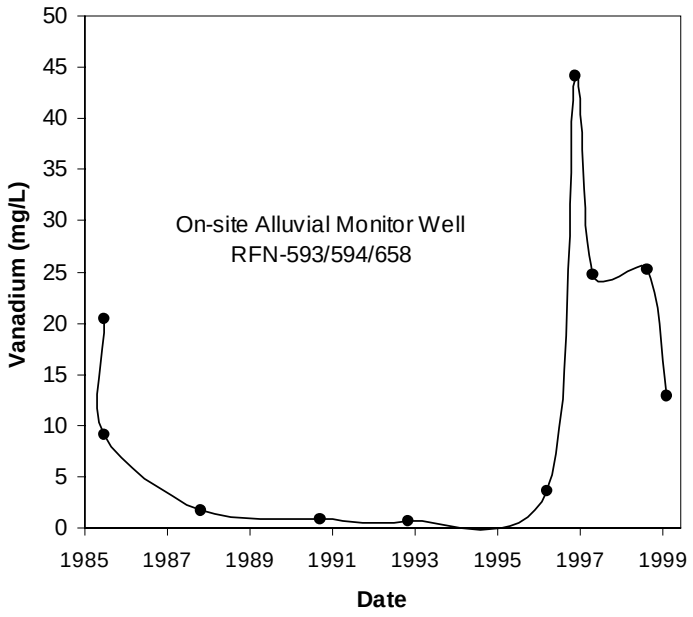

Figure 5–38. Vanadium Concentrations versus Time at On-Site Monitor Well Located near Former Gypsum/Vanadium Evaporation Pond

Concentrations of most constituents in Wasatch background ground water are lower than those found in alluvial background ground water. The exceptions are chloride, sodium, molybdenum, fluoride, and Ra-226. The average concentration of molybdenum in Wasatch ground water exceeds the UMTRA standard of 0.1 mg/L and is significantly higher than molybdenum concentrations detected in the alluvial aquifer. Other commonly detected trace constituents include arsenic, iron, manganese, and uranium. Most of the remaining trace constituents, including antimony, cadmium, lead, and selenium, occur in concentrations below the analytical detection limit. Historical concentrations of background Wasatch ground water have been as high as 0.34 mg/L for selenium, 0.57 mg/L for uranium, and 0.26 mg/L for vanadium (DOE 1996d). Nitrate values as high as 11 mg/L have also been reported (DOE 1996d).

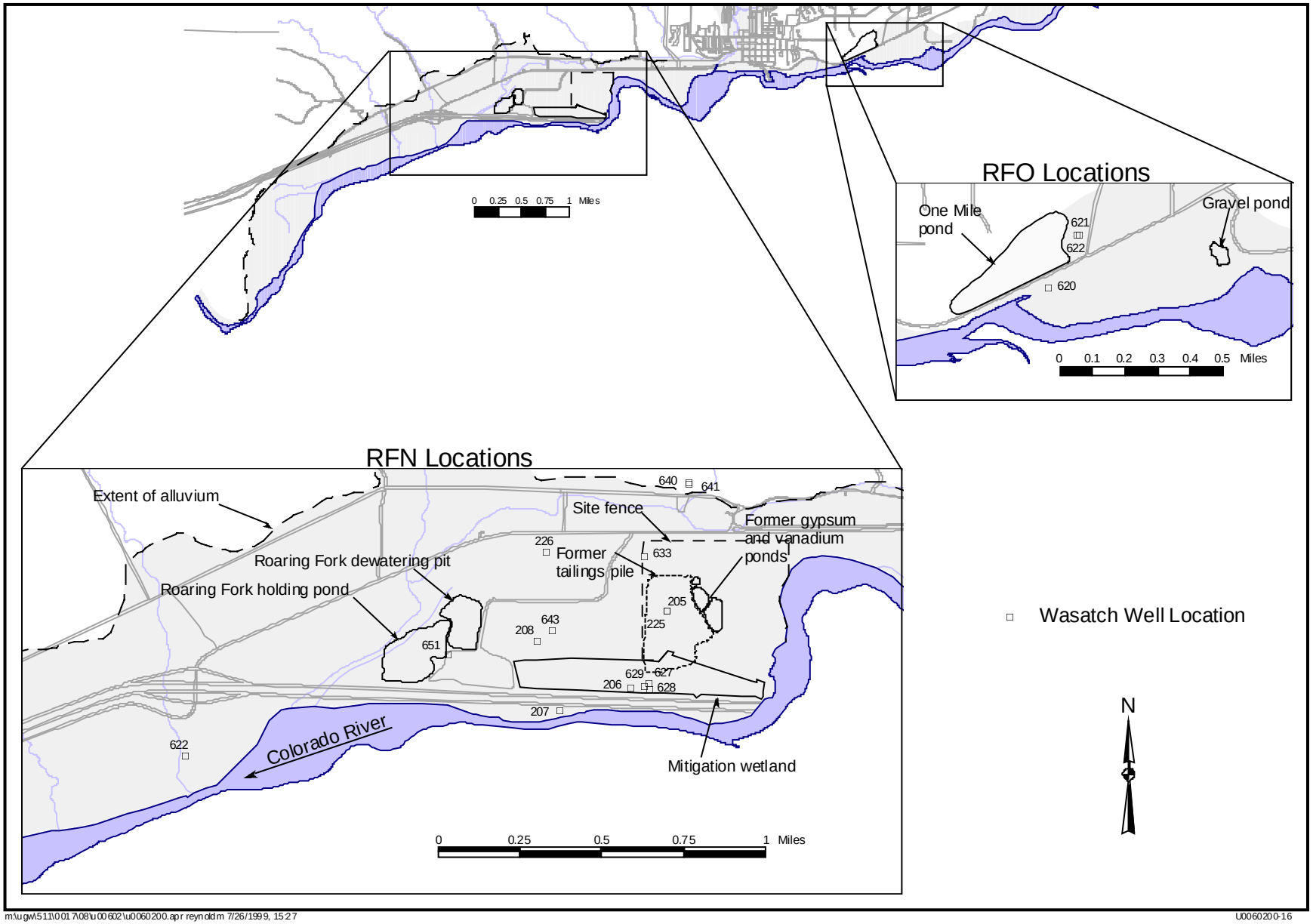

Figure 5–39. Wasatch Monitor Well Locations

Table 5–9. Background Water Quality for the Wasatch Formation—May 1998 through January 1999

Analyte[a]	Unit	Wasatch Background		
		FOD	Mean[b]	Range
Major				
Ammonia as NH_4	mg/L	7/7	.3994	.126-.797
Calcium	mg/L	7/7	64.72	3.53-281
Chloride	mg/L	7/7	913	121-2760
Magnesium	mg/L	7/7	24.92	.873-117
Nitrate	mg/L	4/7	.2437	.0493-.674
Sodium	mg/L	7/7	878.9	443-1810
Sulfate	mg/L	7/7	554	33.7-1350
Metal				
Antimony	mg/L	0/7		<.001-<.001
Arsenic	mg/L	5/7	.0039	<.001-.0064
Cadmium	mg/L	0/7		<.001-<.001
Iron	mg/L	5/7	.05867	<.003-.186
Lead	mg/L	0/7		<.001-<.001
Manganese	mg/L	7/7	.1809	.0043-1.04
Molybdenum	mg/L	7/7	.1625	.0102-.234
Selenium	mg/L	0/7		<.001-<.001
Uranium	mg/L	7/7	.00621	.0013-.0244
Vanadium	mg/L	1/7		<.001-<.002
Other				
Alkalinity as $CaCO_3$	mg/L	7/7	318.9	255-403
Fluoride	mg/L	7/7	2.134	.237-4.01
Redox Potential	mV	7/7	52.7	-65-123
Specific Conductance	umhos/cm	7/7	3725	2170-6310
Sulfate/Chloride		7/7	1.196	.04035-3.849
pH	s.u.	7/7	8.062	7.04-8.54
Radioactive				
Gross Alpha	pCi/L	2/7		<10.74-<49.34
Gross Beta	pCi/L	0/7		<10.97-<48.48
Lead-210	pCi/L	0/7		<.87-<1.3
Polonium-210	pCi/L	2/7		<.06-.16
Radium-226	pCi/L	5/7	.148	<.1-.35
Radium-228	pCi/L	0/7		<.66-<.75
Thorium-230	pCi/L	1/7		<.8-<1.6

[a]All measurements are filtered except specific conductance, pH, redox potential, and alkalinity (as $CaCO_3$) which are unfiltered.
[b]**One-half the detection limit was used to calculate the mean; mean was not calculated if fewer than half of the samples were above detection**

Time-concentration plots for selected constituents in Wasatch background wells are shown in Figures 5–40 through 5–44. As with the alluvial background wells, concentrations vary significantly for some wells over time; there is also a large degree of difference in concentrations between different wells. Wells showing the least variability generally contain constituents present at very low concentration levels. No readily discernible patterns or trends can be identified.

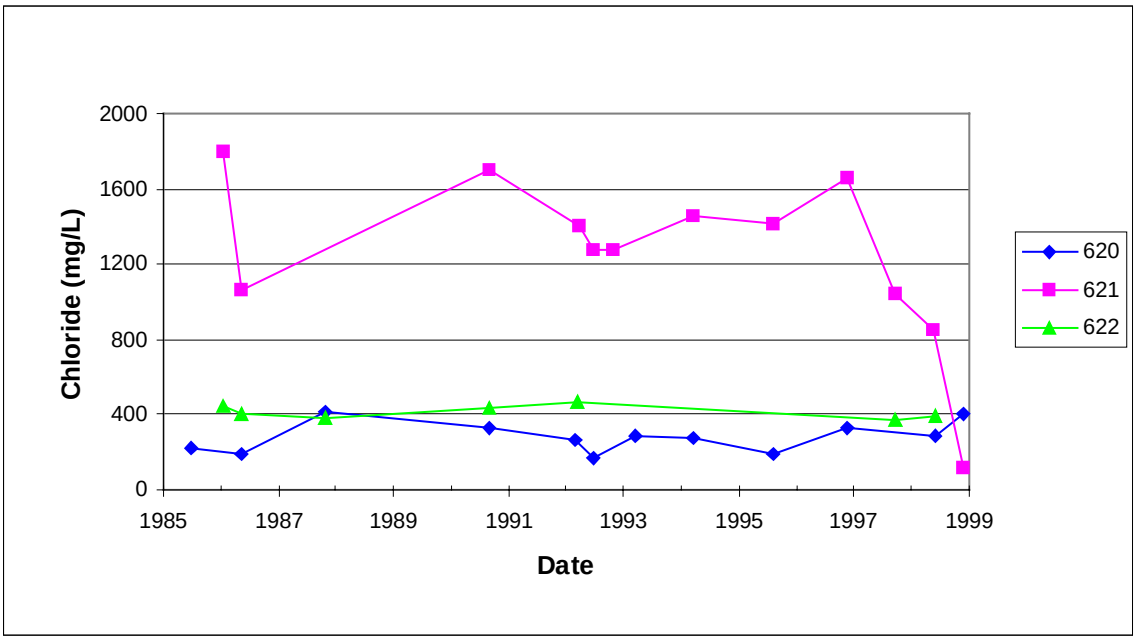

Figure 5–40. Time Concentration Plot for Background Wasatch Ground Water—Chloride

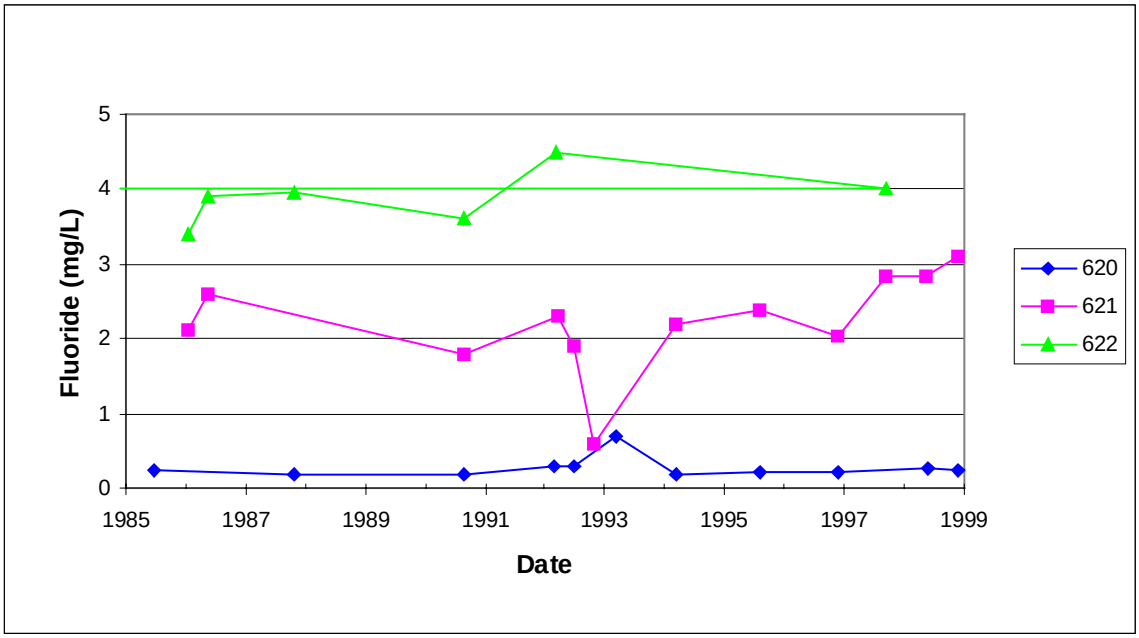

Figure 5–41. Time Concentration Plot for Background Wasatch Ground Water—Fluoride

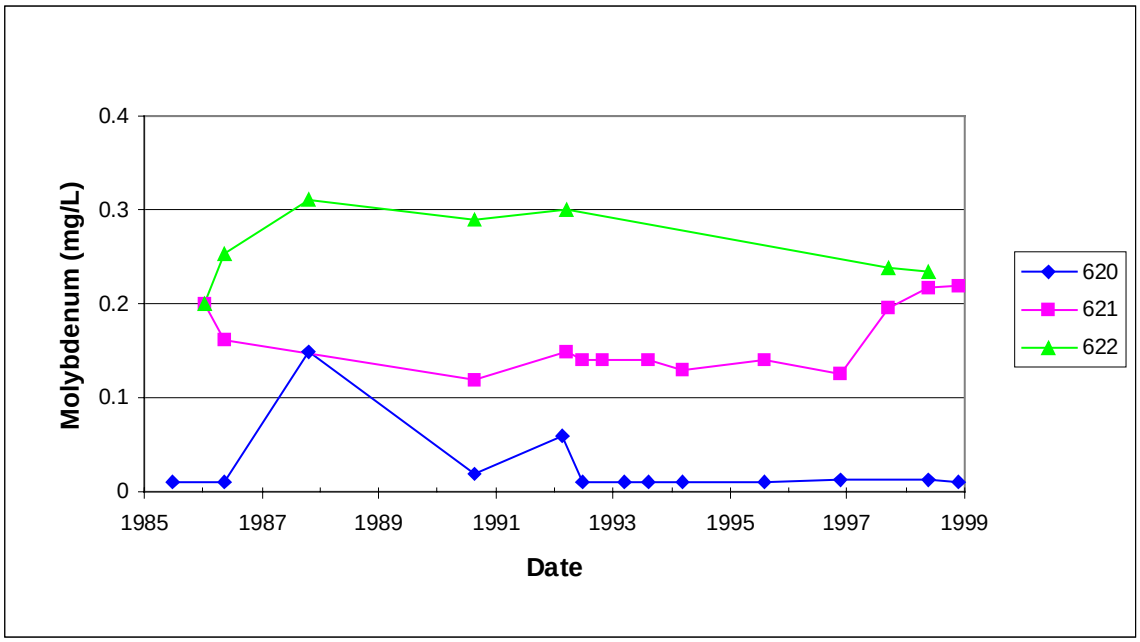

Figure 5–42. Time Concentration Plots for Background Wasatch Ground Water—Molybdenum

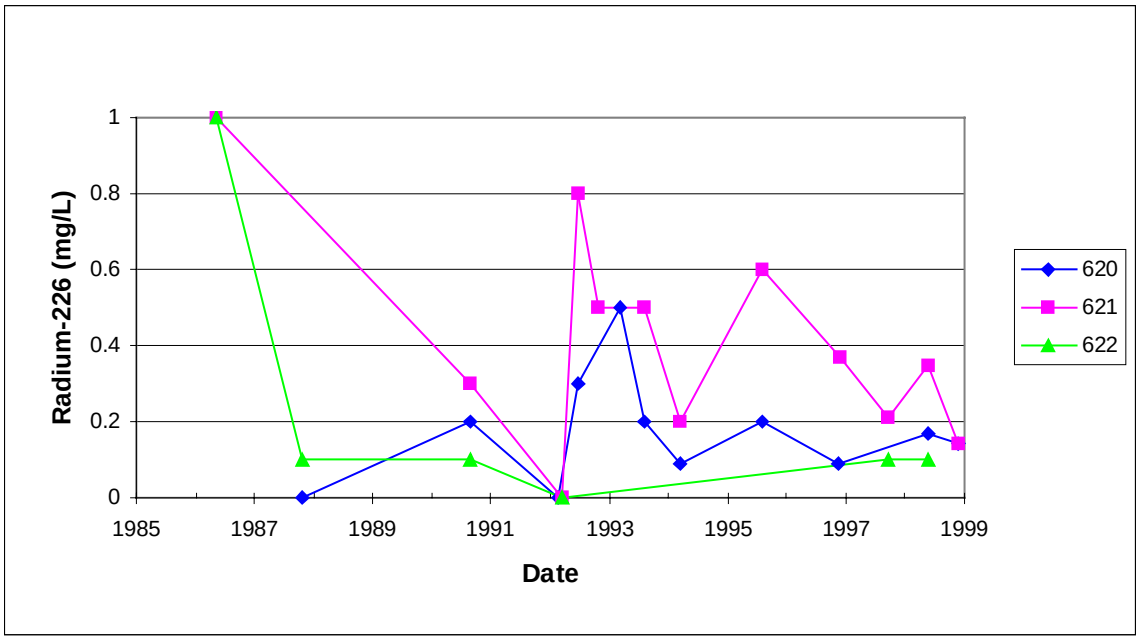

Figure 5–43. Time Concentration Plot for Background Wasatch Ground Water—Radium-226

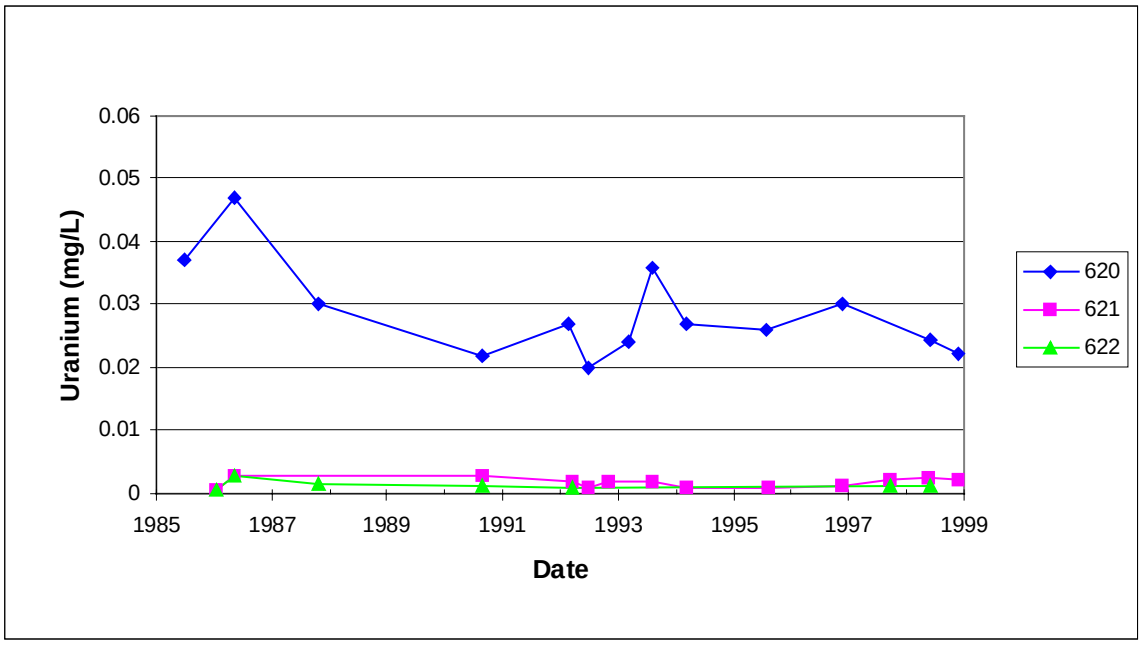

Figure 5–44. Time Concentration Plots for Background Wasatch Ground Water—Uranium

Background water in the Wasatch Formation is characterized by a predominance of the sodium cation (Figure 5–27). The dominant anionic species is either sulfate or chloride. Sodium concentrations range from 443 to 1,810 mg/L, sulfate from 34 to 1,350 mg/L, and chloride from 121 to 2,760 mg/L. Alkalinity, as calcium carbonate, ranges from 255 to 403 mg/L. On average, the Wasatch ground water pH is above neutral (8.1 s.u.) and the redox condition is slightly oxidizing (oxidation-reduction potential 53 mV).

Geochemical modeling of ground water from the Wasatch monitor well RFO–621, using the PHREEQE computer software program (Parkhurst 1980), indicates the background water is at or near saturation with respect to calcite (calcium carbonate), magnesite (magnesium carbonate), and fluorite (calcium fluoride). Equilibrium with fluorite is consistent with observed concentrations of fluoride as high as 4.0 mg/L (Table 5–9).

5.3.4.2 Site Impacts on the Wasatch Formation

Ground water samples were collected in August 1998 and January 1999 at 13 on-site and downgradient monitor wells completed in the Wasatch Formation. Sampling results are summarized in Table 5–10 and presented in Appendix C. The range in natural background is provided in Table 5–10 for comparison. Monitor well locations are shown in Figure 5–39.

Table 5–10. On-Site and Downgradient Wasatch Ground Water at New Rifle —August 1998 and January 1999

Analyte[a]	Unit	Wasatch Aquifer				Background Range
		FOD	Mean[b]	Range	Max Well	
Major						
Ammonia as NH_4	mg/L	26/26	58.12	.186-618	0206	.126-.797
Calcium	mg/L	25/25	183.6	6.74-484	0226	3.53-281
Chloride	mg/L	26/26	739.5	63.7-1470	0205	121-2760
Magnesium	mg/L	25/25	63.81	1.64-210	0206	.873-117
Nitrate	mg/L	23/26	67.13	<.011-964	0643	.0493-.674
Sodium	mg/L	25/25	1237	371-2900	0629	443-1810
Sulfate	mg/L	26/26	2241	27.2-6540	0627	33.7-1350
Metal						
Antimony	mg/L	6/25		<.001-.004	0205	<.001-<.001
Arsenic	mg/L	23/25	.004851	.00062-.0231	0205	<.001-.0064
Cadmium	mg/L	2/25		<.001-.0011	0633	<.001-<.001
Iron	mg/L	12/25		<.003-4.34	0206	<.003-.186
Lead	mg/L	0/25		<.001-<.001	0651	<.001-<.001
Manganese	mg/L	24/25	.4906	<.001-2.29	0206	.0043-1.04
Molybdenum	mg/L	25/25	.173	.0138-.898	0208	.0102-.234
Selenium	mg/L	15/25	.002155	<.0001-.0145	0643	<.001-<.001
Uranium	mg/L	18/25	.0145	<.001-.106	0226	.0013-.0244
Vanadium	mg/L	10/25		<.001-.0106	0225	<.001-<.002
Other						
Alkalinity as $CaCO_3$	mg/L	23/23	277	65-491	0629	255-403
Fluoride	mg/L	26/26	1.129	.101-2.83	0633	.237-4.01
Redox Potential	mV	26/26	7.615	-214-214	0205	-65-123
Specific Conductance	umhos/cm	26/26	6053	2360-12490	0627	2170-6310
Sulfate/Chloride		26/26	3.59	.02612-9.185	0627	.04035-3.849
pH	s.u.	26/26	7.908	6.37-11.12	0225	7.04-8.54
Radioactive						
Gross Alpha	pCi/L	2/25		<11.08-104.9	0226	<10.74-<49.34
Gross Beta	pCi/L	2/25		14.54-32.08	0226	
Lead-210	pCi/L	0/25		<11.5-<96.61	0206	<10.97-<48.48
Polonium-210	pCi/L	0/25		<.87-<1.02	0207	<.87-<1.3
Radium-226	pCi/L	1/25		<.06-.15	0622	<.06-.16
Radium-228	pCi/L	11/25		<.11-1.94	0205	<.1-.35
Thorium-230	pCi/L	18/25	.9687	<.6-1.84	0629	<.66-<.75
		0/25		<.8-<1.3	0651	<.8-<1.6

[a]All measurements are filtered except specific conductance, pH, redox potential, and alkalinity (as $CaCO_3$) which are unfiltered.
[b]One-half the detection limit was used to calculate the mean; mean was not calculated if fewer than half of the samples were above detection

Nitrate, molybdenum, uranium, and gross alpha are the only regulated constituents that exceed an UMTRA ground water standard in the Wasatch Formation. Nitrate concentrations greater than the 44 mg/L standard were detected in ground water collected at monitor wells RFN–208 and –643. Both of these monitor wells are located within 250 ft of each other approximately one-half mile west of the site boundary. Monitor well RFN–208 is constructed with a 10 ft screen beginning approximately 7 ft below the top of the Wasatch contact. Monitor well RFN–643 is installed with a 10-ft screen beginning approximately 37 ft below the Wasatch contact. Both monitor wells are screened across fine-grained sandstone. The maximum nitrate concentration of 964 mg/L was detected in January 1999 in monitor well RFN–643.

Molybdenum was detected above the 0.1 mg/L UMTRA standard in ground water samples collected at monitor wells RFN–207, –208, –226, –622, and –633. Three of the wells, RFN–207, –208, and –226, are completed with 10-ft screens beginning 6 to 8 ft below the contact between the alluvium and Wasatch Formation. Monitor well RFN–622 is completed with a 5-ft screen beginning 10 ft below the Wasatch contact and monitor well RFN–633 is completed with a 10-ft screen approximately 33 ft below the Wasatch contact. The maximum molybdenum concentration of 0.898 mg/L was detected in ground water sampled from the shallow monitor well RFN–208. The standard is also exceeded by the 0.2 mg/L molybdenum concentration observed in the upper range of natural background water samples (Table 5–9). Historically, molybdenum concentrations have been reported above 0.3 mg/L at background monitor well RFO–622 (Figure 5–42). Thus, some of the source of molybdenum observed in the Wasatch is most likely naturally occurring.

Uranium was detected in ground water samples at concentrations slightly above the 0.044 mg/L UMTRA standard at monitor wells RFN–208 and –226. Uranium concentrations observed in water sampled from monitor well RFN–208 ranged from 0.056 mg/L in August 1998 to 0.061 mg/L in January 1999, values that are only slightly above the standard. The maximum uranium concentration of 0.106 mg/L was detected in ground water sampled at monitor well RFN–226 in August 1998. The uranium concentration at this location decreased to 0.078 mg/L in January 1999.

Several other constituents in ground water samples collected from Wasatch monitor wells exceed the upper range in natural background. However, most of these constituents, including calcium, chloride, magnesium, sodium, sulfate, and iron can be attributed to natural variations within the Wasatch Formation rather than downward vertical contaminant migration from the alluvial plume. One exception is ammonia, which exceeds the upper range in natural background at several Wasatch monitor wells. The most notable exceedance is at monitor well RFN–206 where a maximum concentration of 618 mg/L was detected in the ground water sample collected January 1999. Monitor well RFN–206 is installed with a 10-ft screen beginning approximately 6 ft below the Wasatch contact. Ammonia, however, does not appear to have migrated to the lower confined and semiconfined flow system as evidenced by ground water samples collected at different depth intervals from several other monitor wells located within a 100-ft radius of RFN–206. Results of ammonia analyses for samples collected in January 1999 are summarized in Table 5–11 in order of increasing screen depth. All these wells are nested downgradient of the former source area, as shown in Figure 5–45, in a location near the center of the mitigation wetland.

Table 5–11. Summary of Contaminant Concentrations at Five Nested Monitor Wells

Monitor Well (RFN)	Completion Zone	Depth (ft) of Top of Screen below Alluvial-Wasatch Contact	Screen length (ft)	Ammonia (mg/L)	Nitrate (mg/L)	Molybdenum (mg/L)	Uranium (mg/L)
217	Alluvium	NA	15	319	0.393	1.54	0.107
206	Wasatch	6	10	618	0.475	0.0684	0.0039
629	Wasatch	16	10	89	0.660	0.0319	0.0061
627	Wasatch	38	20	101	0.630	0.0403	0.0059
628	Wasatch	63	20	0.9	0.426	0.0237	0.0010

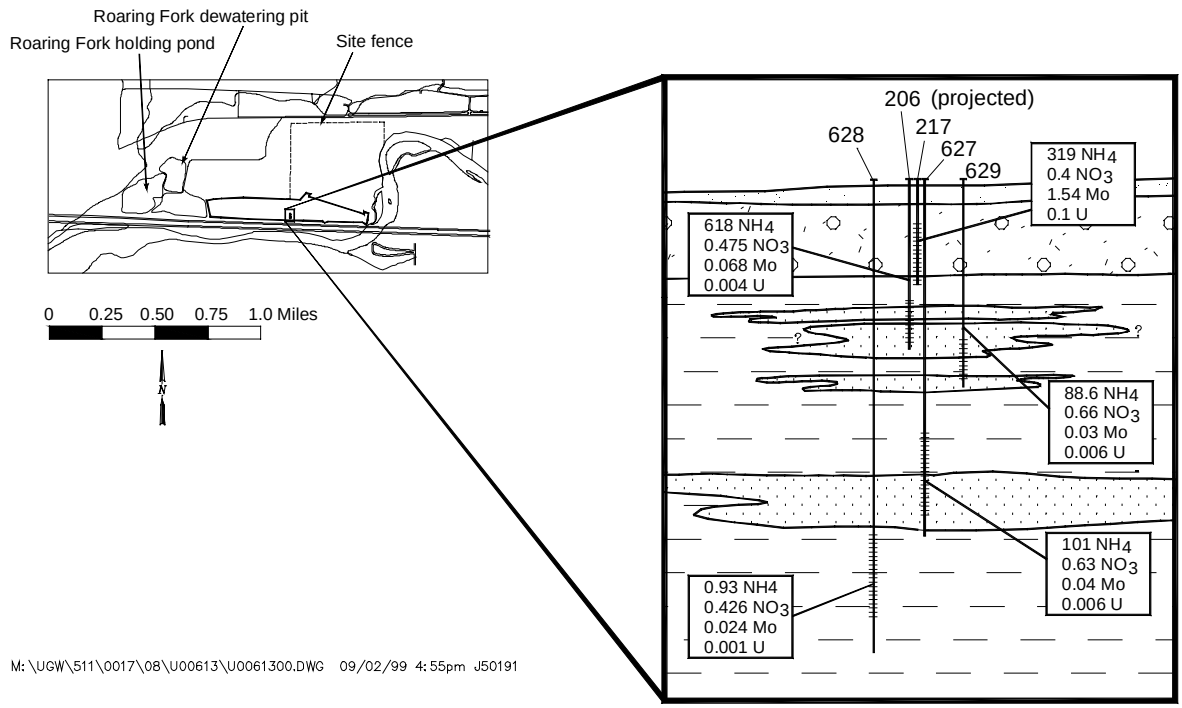

Figure 5–45. Ground Water Concentrations (mg/L) of Selected Site-related Contaminants at Five Nested Monitor Well Locations

Ammonia concentration of 319 mg/L is observed in the overlying alluvial plume water sampled from monitor well RFN–217. However, the ammonia concentration increases to 618 mg/L in ground water sampled from the shallowest Wasatch well RFN–206. This suggests the ammonia concentration in the alluvial plume has historically been much higher and has since migrated further downgradient (Section 5.3.3.2). Concentrations in the Wasatch appear to decrease rapidly with depth as evidenced by the relatively low ammonia concentration of 0.9 mg/L observed in the deepest completion at RFN–628. The relatively low-permeability claystone and siltstone beds of the Shire Member form aquitards that separate the overlying alluvial aquifer from the deeper and more permeable sandstone of the middle Molina Member; thereby reducing the possibility of downward contaminant migration to the lower aquifer.

Similar results are observed for nitrate, molybdenum, and uranium. For nitrate, the plume in the overlying alluvial aquifer has migrated a significant distance away from the former source area as evidenced by the distribution pattern revealed in Figure 5–32 and by the background nitrate concentration of 0.393 mg/L measured in the alluvial water at RFN–217. Conversely, molybdenum and uranium concentrations in the alluvial plume water collected at monitor well RFN–217 are elevated above their respective UMTRA ground water standard. However, the concentrations for all three constituents are within the range in natural background for all the ground water samples that were collected from these nested Wasatch wells. Complex vertical gradients that exist in the Shire Member are the result of aquitard forming claystone and siltstone units that separate alluvial ground water from the lower flow system in the Molina Member (Section 5.2.3). These aquitards in the Shire Member limit downward contaminant migration from the alluvium to the lower flow system. The absence of elevated ammonia, nitrate,

molybdenum, and uranium observed in the ground water samples collected from the deeper Wasatch well RFN–628, which is also completed in the Shire Member, provides a good indication that the lower flow system in the Molina Member has not been negatively impacted by site-related contamination.

5.3.5 Contaminant Fate and Transport

Ground water discharge to the Colorado River and geochemical processes will eventually decrease the contaminant concentrations in the alluvial aquifer at the New Rifle site to background levels. The time required for one pore volume of ground water to migrate from the most contaminated upgradient location on the site to a point of discharge into the Colorado River approximately 5,000 ft southwest of the site is estimated at 11 years assuming 1.25 ft/day as the average linear ground water flow rate. However, as contaminated ground water migrates downgradient, some of the contaminant is subject to dispersion as the water moves through the heterogeneity of the aquifer. In addition to dispersion, as contaminated ground water migrates through the alluvial aquifer, some of the contamination transfers between the solid and liquid phases. These geochemical phenomena cause the contamination to travel at a slower rate than the average ground water velocity. Therefore, the time required for contaminants in the alluvial system to naturally flush clean can be significantly greater than 11 years and one pore volume of ground water.

The geochemical processes that cause contaminant retardation can include adsorption, absorption, precipitation, diffusion into immobile porosity, transfer to vapor phases, and so on (these collective processes will simply be referred to as "adsorption" in the rest of this discussion). It is generally not possible to differentiate among all of these processes. However, for many aquifer systems, a bulk parameter (the distribution coefficient or K_d) has been used with some success to describe the retardation of contamination. Single-point K_d values were determined for As, Mo, Se, U, and V (Section 4.0). A summary of the results is presented in Table 5–12.

Table 5–12. Summary of Distribution Coefficients in Alluvium at the New Rifle Site

Parameter	K_d (mL/g)				
	As	Mo	Se	U	V
Mean	5.5	0.1	2.3	0.7	4.9
Minimum	1.2	-0.2	1.1	-0.3	1.2
Maximum	9.2	0.4	4.2	1.4	10.5

The average K_d values presented in Table 5–12 indicate that As, Se, and V are likely to sorb to soils in the subsurface, while Mo and U are not. In addition, the range of K_d values is highly variable for As, Se, and V. The negative values observed for Mo and U suggest that even background levels in the subsurface could be contributing to levels detected in ground water. In general, the observed K_d values at the New Rifle site indicate that movement in alluvial aquifer ground water is expected to be retarded for V and As, and to a lesser extent for Se, through adsorption by aquifer materials. Conversely, Mo and U are expected to be conserved in the ground water, as evidenced by average K_d values less than 1, which indicate little or no retardation. The relative mobility of As, Mo, Se, U, and V is consistent with the contaminant dispersion patterns observed in ground water samples collected from the most recent sampling in 1998 (Figures 5–30, –31, –35, –36, and –37).

A quantitative computer model was developed to predict the concentration and distribution of the contaminants of concern (COCs) in the future (Appendix D). The model incorporates the various flow and transport parameters that affect the hydraulic head and contaminant distribution.

5.3.5.1 Computer Flow and Transport Model

Natural flushing is an acceptable remediation strategy that allows natural processes to reduce the ground water contaminants to background levels, MCLs, or ACLs provided that: (1) contaminant concentrations decrease to the regulatory limits within 100 years, (2) institutional controls to protect the public health and environment are maintained during the cleanup period, and (3) the ground water is not currently, and is not projected to become, a source for a public water system (Section 2.0). A ground water flow and transport model was developed to evaluate if natural processes will reduce site-related concentrations of As, Mo, NO_3, Se, U, and V to regulatory levels in the alluvial aquifer within the 100-year time frame. Vanadium was selected for modeling because of its extremely high levels compared to background. All other modeled constituents are regulated by UMTRA.

The existing ground water flow pattern at the New Rifle site was modeled with the MODFLOW software package (McDonald and Harbaugh 1988), a multi-layered, three-dimensional hydrologic flow model published by the USGS. Results of the flow model were then coupled to MT3D (Zheng 1990), a modular three-dimensional transport model to simulate advection and dispersion of contaminants in the ground water system. Hydraulic conductivity, dispersivity, and K_d are identified as the most sensitive input parameters to the MODFLOW and MT3D simulations. By varying these sensitive input parameters during each of the 100 simulations that were performed to predict the mean concentration, the variance associated with the mean predicted concentration was used to calculate the probability that the contaminant concentration will exceed the respective cleanup goal.

Initial MT3D modeling results indicate the Roaring Fork dewatering activities enhance the natural flushing of contaminants. In general, the dewatering activities tend to accelerate migration of the contaminant plume beneath the site toward the dewatering pit. Additionally, the ground water mound created by the holding pond creates a radial warping of flow directions downgradient from the gravel operations which tends to disperse and dilute contaminant concentrations to a greater extend than natural flushing alone. It appears from the initial modeling results that the dewatering activities may shorten the natural flushing time period required to met the regulatory standard for a conservative species such as uranium by as much as 10 percent assuming a 10-year economic life-span for the gravel reserves. For purposes of evaluating if the natural flushing option is acceptable as a compliance strategy, the MT3D modeling was performed simulating natural flushing with no dewatering activities. This results in a more conservative estimate of predicted ground water concentrations.

Details regarding the model's construction, calibration, sensitivity analysis, and stochastic parameters are presented in Appendix D. These codes are fully described in the references cited and have been verified, benchmarked, and approved for use by most government and regulatory agencies. A summary of the modeling results is presented in the following section.

Natural Flushing Arsenic

Arsenic concentrations above the 0.05 mg/L UMTRA ground water standard are associated with a relatively small ground water plume beneath the center of the former gypsum-vanadium evaporation pond as previously illustrated by the plume map presented in Figures 5–29 and 5–30. The maximum detected concentration of arsenic for the August 1998 sampling event was 0.30 mg/L. In January 1999 this concentration decreased to 0.08 mg/L, a value only slightly above the cleanup goal. The higher 1998 value of 0.30 mg/L is a conservative estimate of the initial concentration for the MT3D contaminant transport modeling. Predicted ground water concentrations above the proposed 0.05 mg/L arsenic standard at 100 years into the future is presented in Figure 5–46. The results indicate that the maximum arsenic concentration is predicted to decrease to 0.06 mg/L after 100 years of natural flushing. Stochastic simulations based on the August 1998 sampling results as the initial concentrations predict a small area near the Colorado River where the probability is greater than 5 percent that the 0.05 mg/L ground water standard will be exceeded at 100 years (Figure 5–47). The probability is less than 5 percent that the arsenic concentrations will exceed the standard in the alluvial aquifer in all other areas of the site.

The contaminant transport simulation was repeated using the most recent January 1999 sampling results for the initial concentration. In 1999 the maximum concentration was 0.08 mg/L, a value only slightly above the ground water standard (Figure 5–30). After 15 years of natural flushing the maximum arsenic concentration is predicted to decrease to 0.048 mg/L which is below the standard. A probability contour map, based on the most recent January 1999 sampling results as the initial concentrations, indicate a very small area beneath the New Rifle site where the probability is 5 to 10 percent that the arsenic concentrations will exceed the 0.05 mg/L ground water standard at 15 years into the future (Figure 5–48). After 20 years of natural flushing the MT3D stochastic simulations predict the probability is less than 5 percent that the arsenic concentrations will exceed 0.05 mg/L in the alluvial aquifer in all areas of the site, effectively meeting the UMTRA ground water standard.

Natural Flushing Molybdenum

Molybdenum concentrations at the New Rifle site exceed the 0.1 mg/L UMTRA ground water standard for the August 1998 sampling event (Figure 5–31). The highest concentrations, ranging between 3.6 and 6.4 mg/L, appear to be located beneath the former tailing area (RFN–218, –219, and –659) and the former gypsum-vanadium evaporation ponds (RFN–658). The plume extends off site in a southwest-west direction as far downgradient as the Roaring Fork gravel operation. Predicted ground water concentrations above the 0.1 mg/L molybdenum standard at 20 years into the future are presented in Figure 5–49. As evident by Figure 5–49, the areal extent of the molybdenum plume is significantly reduced through natural flushing. After 25 years of natural flushing the maximum molybdenum concentration is predicted to decrease to 0.048 mg/L which is less than the 0.1 mg/L standard. MT3D stochastic simulations indicate a probability less than 5 percent that the standard will be exceeded in the alluvial aquifer 25 years into the future.

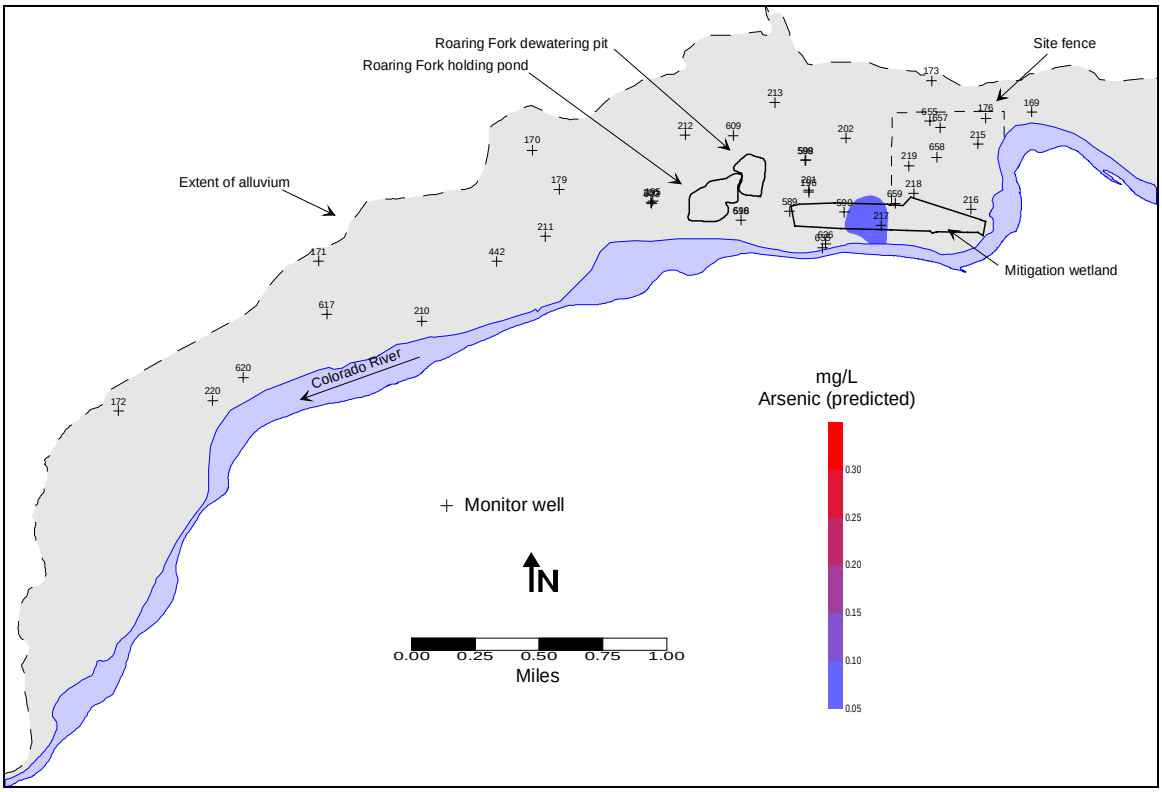

Figure 5–46. Predicted Arsenic Concentrations in the Ground Water at the New Rifle Site at 100 Years Based on the August 1998 Sampling Data—100 Simulations

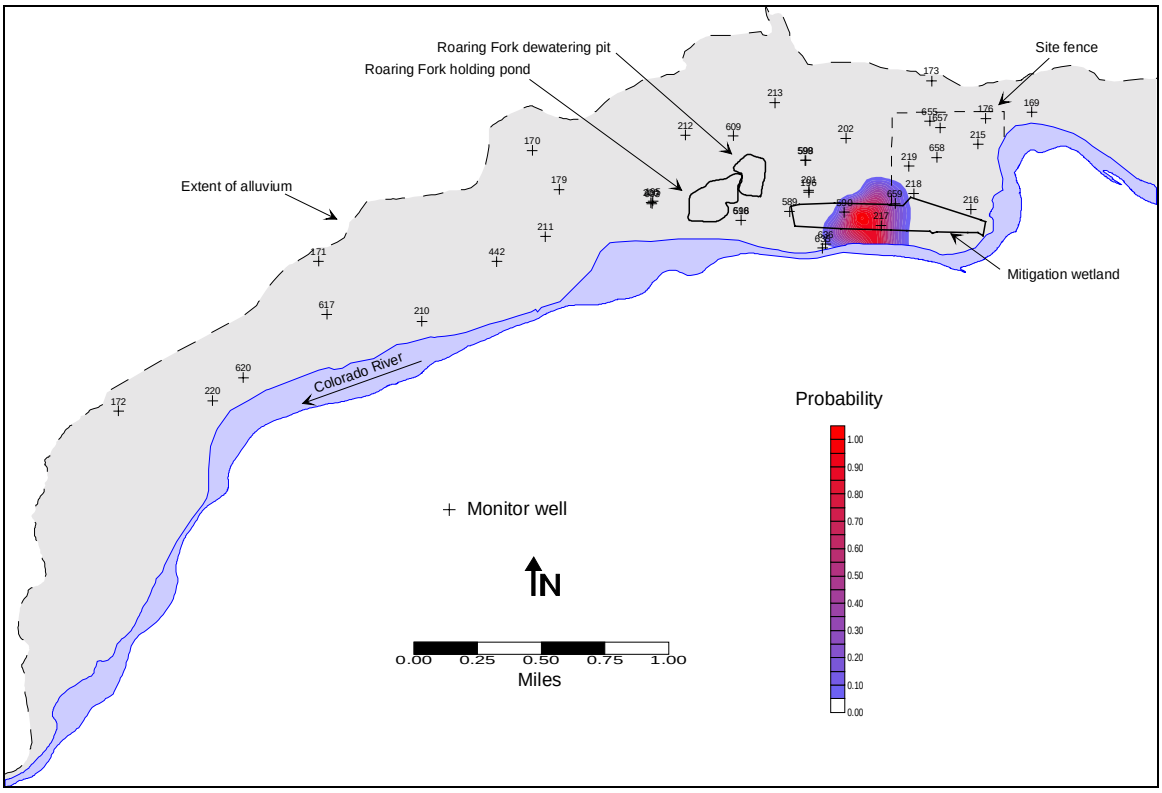

Figure 5–47. Probability of Arsenic Concentrations Exceeding the 0.05 mg/L UMTRA Ground Water Standard at the New Rifle Site at 100 Years Based on the August 1998 Sampling Data—100 Simulations

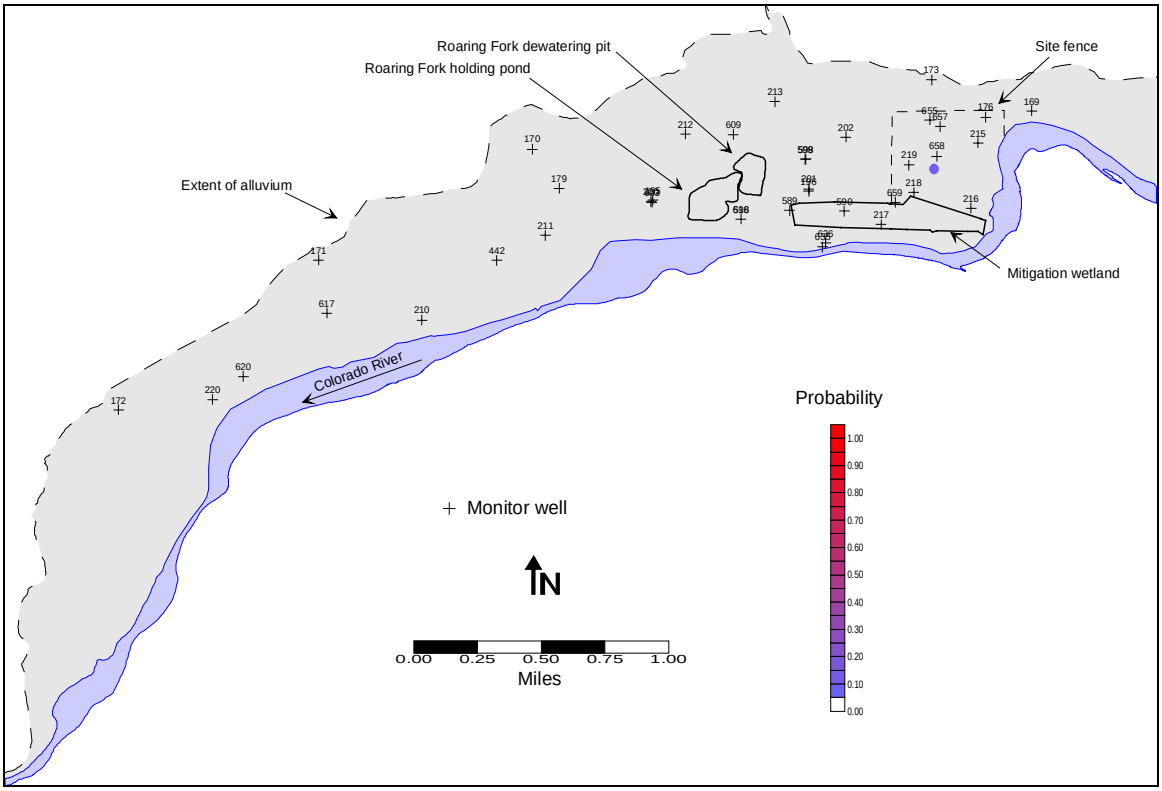

Figure 5–48. Probability of Arsenic Concentrations Exceeding the 0.05 mg/L UMTRA Ground Water Standard at the New Rifle Site at 15 Years Based on the January 1999 Sampling Data—100 Simulations

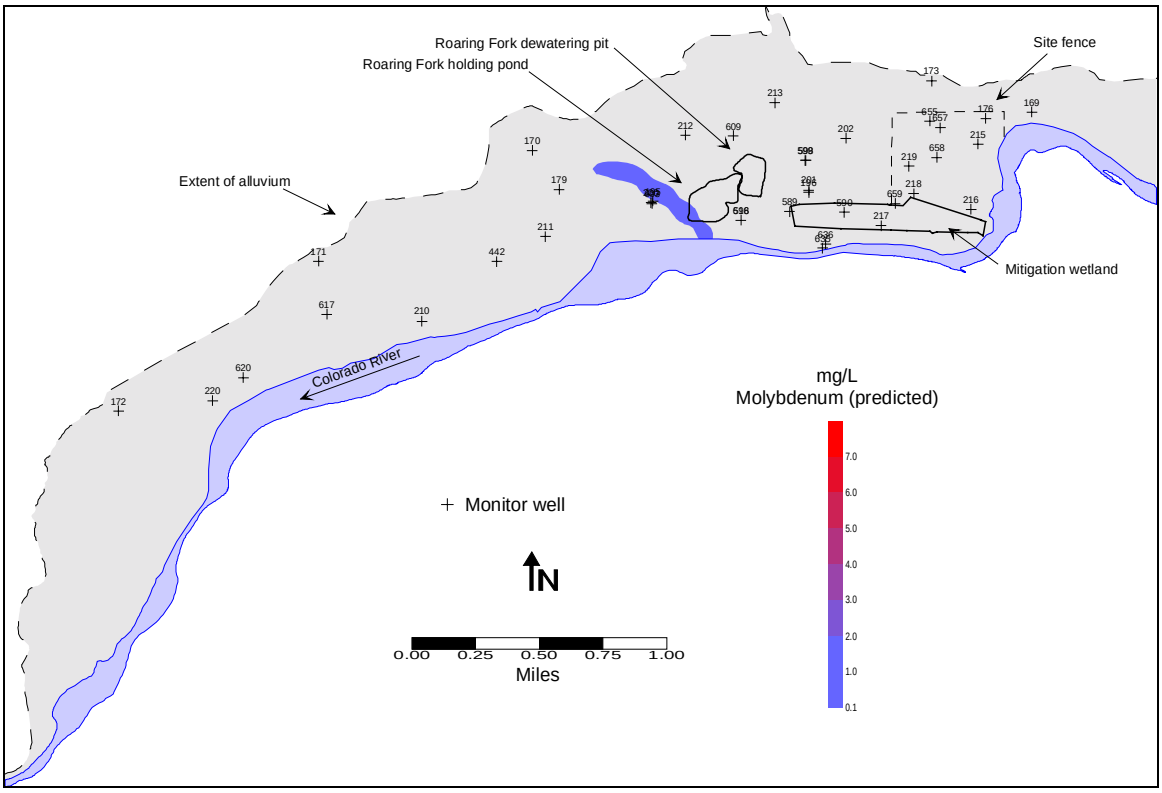

Figure 5–49. Predicted Molybdenum Concentrations in the Ground Water at the New Rifle Site at 20 Years Based on the August 1998 Sampling Data—100 Simulations

Natural Flushing Nitrate

Geochemical and biological reactions are believed to be the primary factors in governing the subsurface behavior of nitrate and associated species. However, biodegradation processes in the subsurface are complex and often do not follow the first-order rate reaction equations (Zheng 1990). In addition, the biodegradation decay constant for nitrate in the dissolved and sorbed phases at the New Rifle site are not known. For these reasons, a linear isotherm (K_d) was used in the MT3D simulations although adsorption is not typically thought to play a role in nitrate fate and transport in the ground water. Results of K_d analysis support this generalization (Calculation Set U0058100). Nitrate K_ds were determined to be equal or less than zero, indicating no affinity for nitrate adsorption to soil. A conservative value of 0.1 mg/L measured for molybdenum (Table 5–12) was used as the K_d for nitrate simulations. Assuming a linear sorption isotherm rather than a biodegradation rate results in a more conservative estimate of the predicted concentrations.

Nitrate concentrations up to 635 mg/L in the alluvial ground water were detected from the August 1998 sampling event at monitor well RFN–589 which is located between the Roaring Fork dewatering pond and the west end of the mitigation wetland. Results of the 1998 sampling and analysis are presented in Figure 5–33 for comparison to the predicted concentrations at 10 years into the future (Figure 5–50). Predicted concentrations above the 44 mg/L UMTRA ground water standard, shown in Figure 5–50, indicate that most of the nitrate has naturally flushed from beneath the site after 10 years. The maximum concentration is approximately 120 mg/L at this time period. After 15 years of natural flushing the maximum concentration decreases to 33 mg/L which is less than the UMTRA ground water standard. Results of the 100 stochastic simulations indicate at 15 years into the future the probability is less than 5 percent that the nitrate concentrations in ground water beneath any area of the site will exceed the UMTRA 44 mg/L standard. Similar results are predicted using the most recent January 1999 sampling and analysis data for the initial concentrations.

Natural Flushing Selenium

Selenium concentrations above the 0.05 mg/L SDWA standard (proposed ACL; see Sections 6.3 and 7.0) are generally confined to the site (Figures 5–34 and 5–35). The highest concentration of 0.78 mg/L selenium was detected at monitor well RFN–658 which is located near the center of the former gypsum-vanadium evaporation pond. In January 1999 this concentration decreased to 0.16 mg/L. The more conservative value of 0.78 mg/L was used as the initial concentration for the predictive modeling. Predicted ground water concentrations above the 0.05 mg/L selenium standard are presented in Figure 5–51. Results indicate the maximum selenium concentration is expected to decrease to 0.08 mg/L after 75 years of natural flushing.

The maximum predicted selenium concentration in the ground water after 100 years of natural flushing is 0.042 mg/L, indicating that on average, the 0.05 mg/L SDWA standard will be met in approximately 100 years. Stochastic simulations using the more conservative August 1998 sampling data as the initial concentrations indicate the there is only a 5 to 28 percent probability that the standard will be exceeded beneath a very small area adjacent to the Colorado River, as the ground water plume naturally flushes for 100 years into the Colorado River (Figure 5–52).

Natural Flushing Uranium

Predicted uranium concentrations above the UMTRA 0.044 mg/L ground water standard at 20 years into the future is presented in Figure 5–53. Uranium concentrations and the relative extent of the plume are predicted to decline significantly from the concentrations observed in August 1998, as seen in Figure 5–36 for comparison. On average, the maximum uranium concentration is predicted to decrease from 0.40 mg/L observed in August 1998 to 0.09 mg/L after 20 years of natural flushing.

After 40 years of natural flushing the MT3D simulations predict that all the site-related uranium will be essentially flushed from the alluvial aquifer. Results of the 100 stochastic simulations indicate at 40 years into the future the probability is less than 5 percent that the uranium concentrations in ground water beneath any area of the site will exceed the UMTRA 0.044 mg/L standard.

Natural Flushing Vanadium

Vanadium concentrations above the proposed 0.33 mg/L cleanup goal (see Sections 6.3 and 7.0) are shown in Figure 5–37 for the August 1998 sampling event. The distribution of elevated vanadium concentrations is mostly confined to the site and to the mitigation wetland located southwest of the site. The maximum vanadium concentration of 25.3 mg/L was detected in the ground water sample collected from beneath the former gypsum-vanadium evaporation pond at monitor well RFN–658. In January 1999 the vanadium concentration at this location decreased to 12.9 mg/L.

Predicted ground water concentrations above the proposed 0.33 mg/L cleanup goal at 100 years into the future are presented in Figure 5–54. As evidenced from Figure 5–54, the model predicts the maximum vanadium concentration will decrease to 4.4 mg/L at 100 years; a value that is above the cleanup goal. It also appears that the areal extent of the plume does not appear to decrease significantly in size after 100 years of natural flushing. The relatively low amount of plume dispersion and concentration dilution predicted after 100 years of natural flushing is consistent with the relatively high K_d measured for vanadium.

Similar results are predicted using the January 1999 sampling data as the initial vanadium concentration (Figure 5–55). For example, the maximum vanadium concentration is predicted to decrease to 2.9 mg/L at 100 years; a value that is also above the cleanup goal. The MT3D model predictions indicate approximately 270 years of natural flushing is required to reduce the January 1999 vanadium concentrations to the 0.33 mg/L cleanup goal. The modeling results also suggest that the initial concentration at monitor well RFN–658 needs to be reduced to approximately 1.25 mg/L for the maximum vanadium concentration to naturally flush to a level below the cleanup goal within 100 years. Therefore, it is apparent from the MT3D modeling results that natural flushing will not achieve the 0.33 mg/L cleanup goal for vanadium within the 100-year compliance time frame. However, it should be noted that the most recent decreases in vanadium observed at monitor well RFN–658 (Figure 5–38) are inconsistent with the relative immobility predicted by the K_d (Section 4.4). If vanadium continues to decrease at the rate observed at historical wells RFN–593 and –594 prior to the surface construction (Figure 5–38) the current vanadium concentrations will be less that 2 mg/L in approximately 4 to 5 years, allowing natural flushing to reduce the remaining vanadium below the cleanup goal within the 100-year time period.

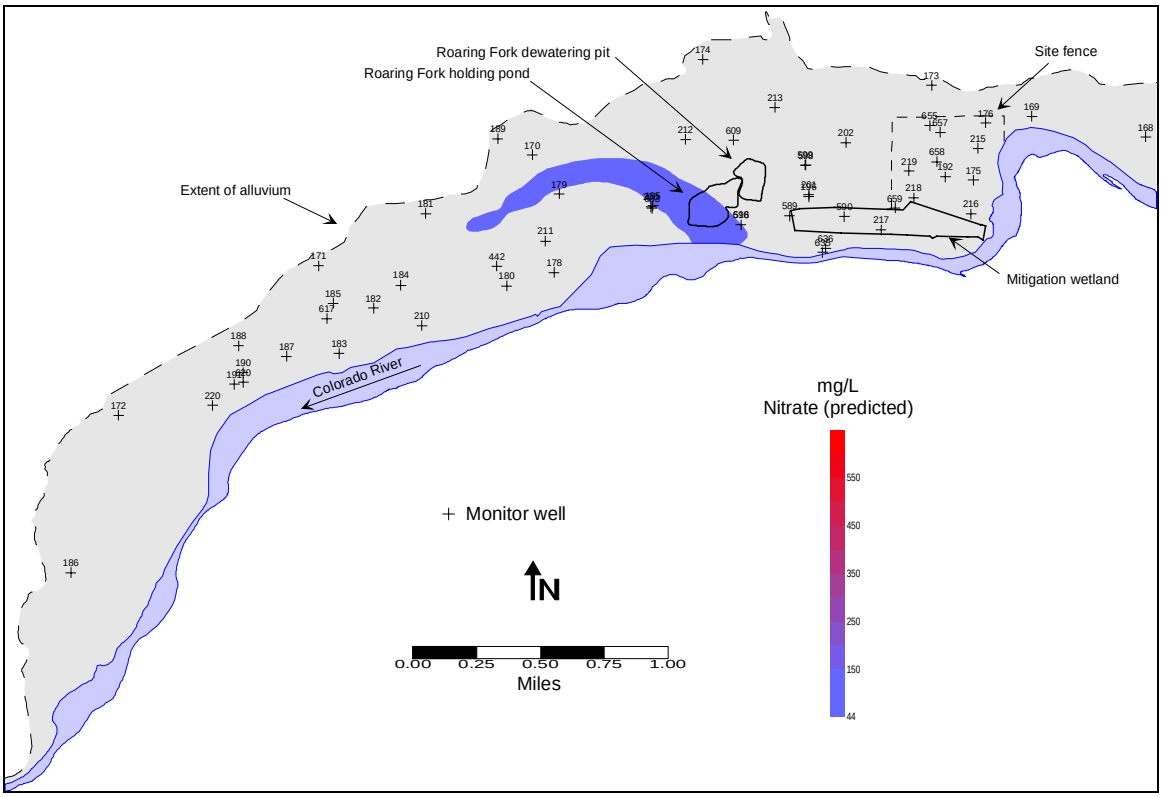

Figure 5–50. Predicted Nitrate Concentrations in the Ground Water at the New Rifle Site at 10 Years Based on the August 1998 Sampling Data—100 Simulations

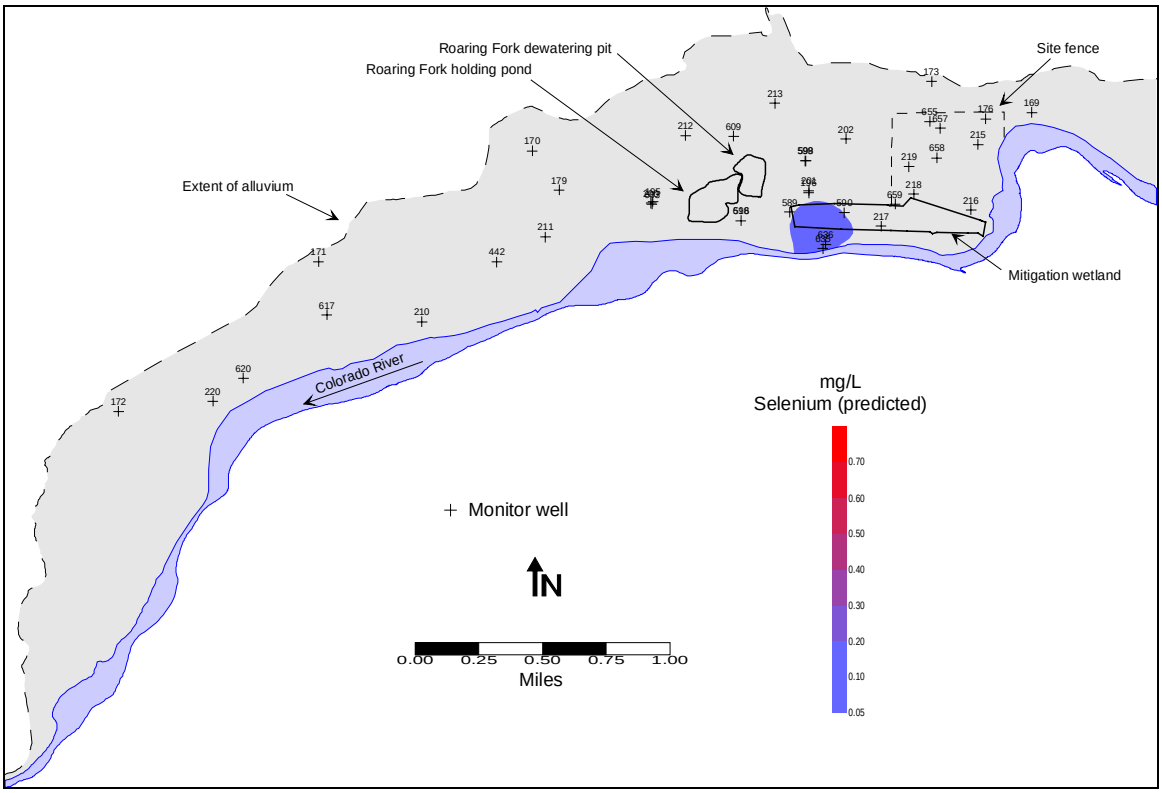

Figure 5–51. Predicted Selenium Concentrations in the Ground Water at the New Rifle Site at 75 Years Based on the August 1998 Sampling Data—100 Simulations

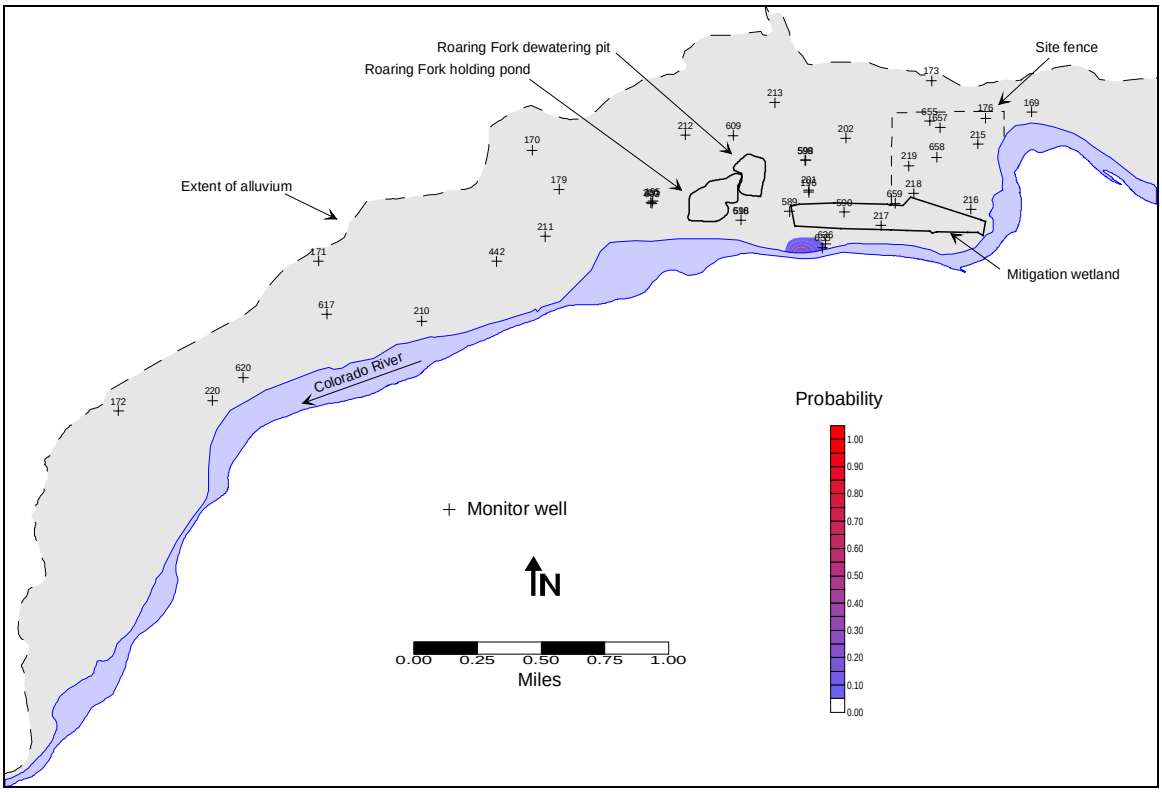

Figure 5–52. Probability of Selenium Concentrations Exceeding the 0.05 mg/L SDWA Ground Water Standard at the New Rifle Site at 100 Years Based on the August 1998 Sampling Data—100 Simulations

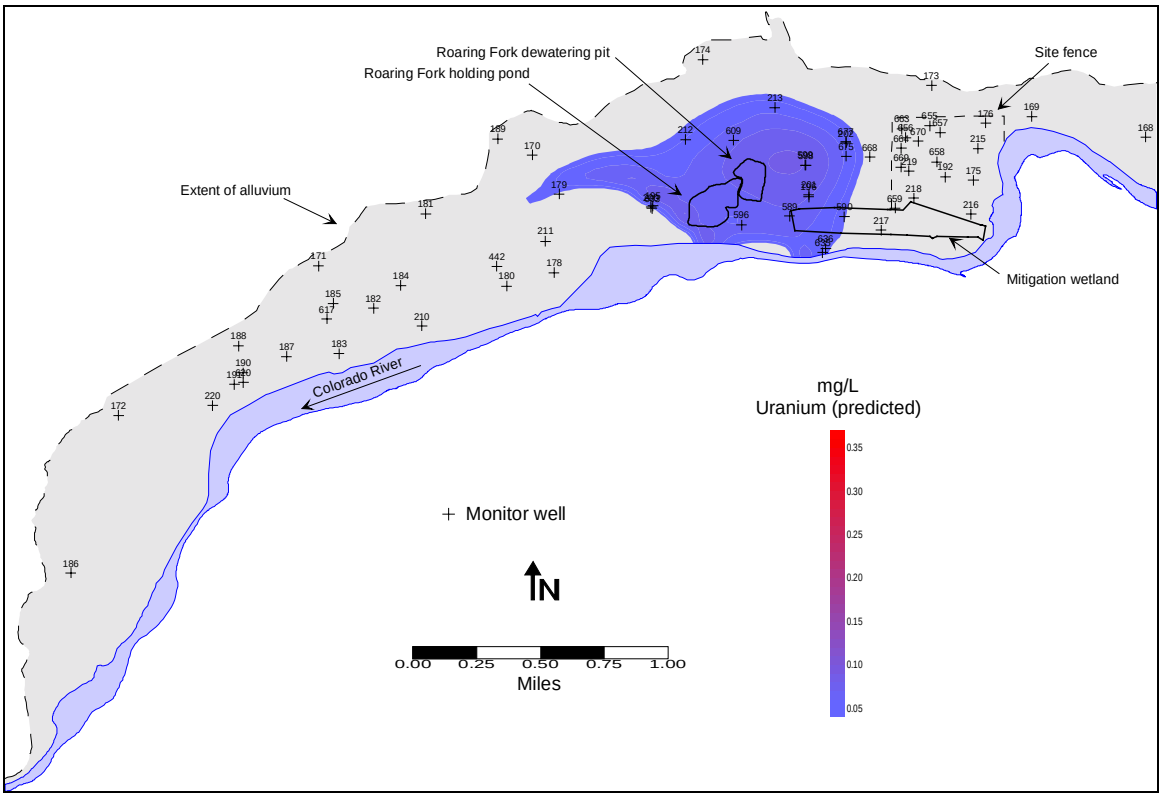

Figure 5–53. Predicted Uranium Concentrations in the Ground Water at the New Rifle Site at 20 Years Based on the August 1998 Sampling Data—100 Simulations

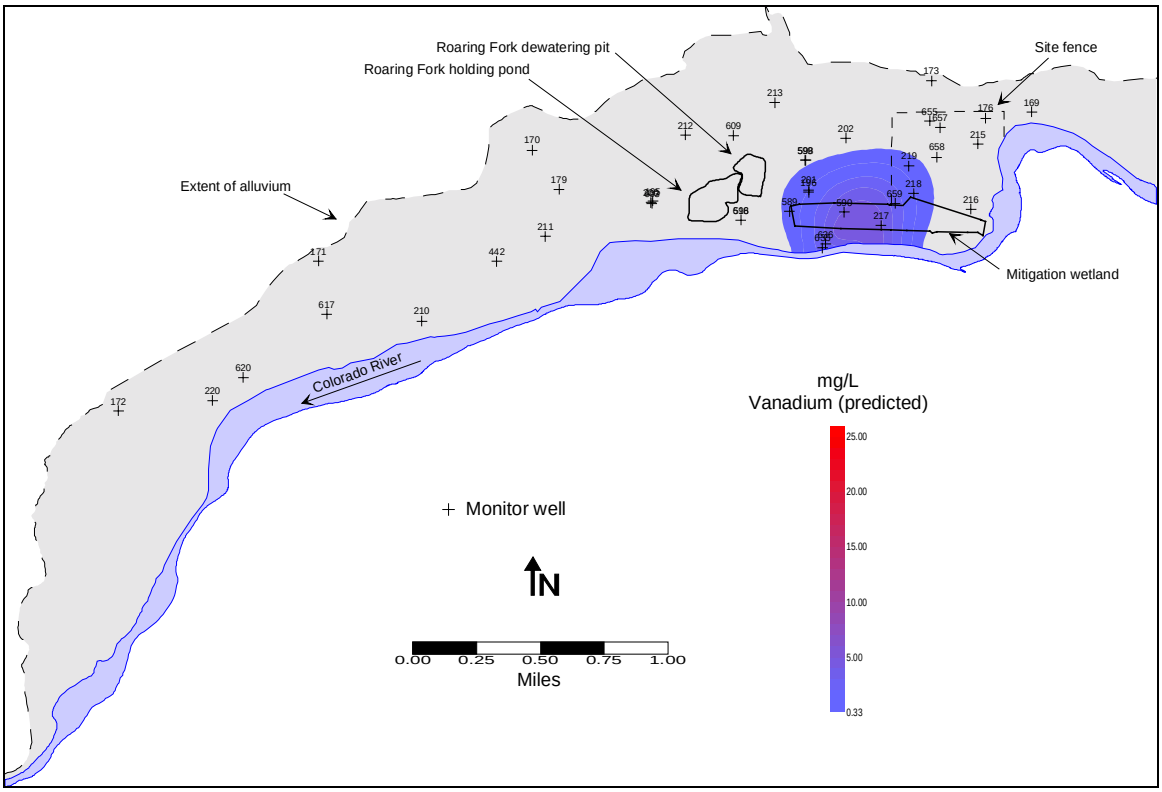

Figure 5–54. Predicted Vanadium Concentrations in the Ground Water at the New Rifle Site at 100 Years Based on the August 1998 Sampling Data—100 Simulations

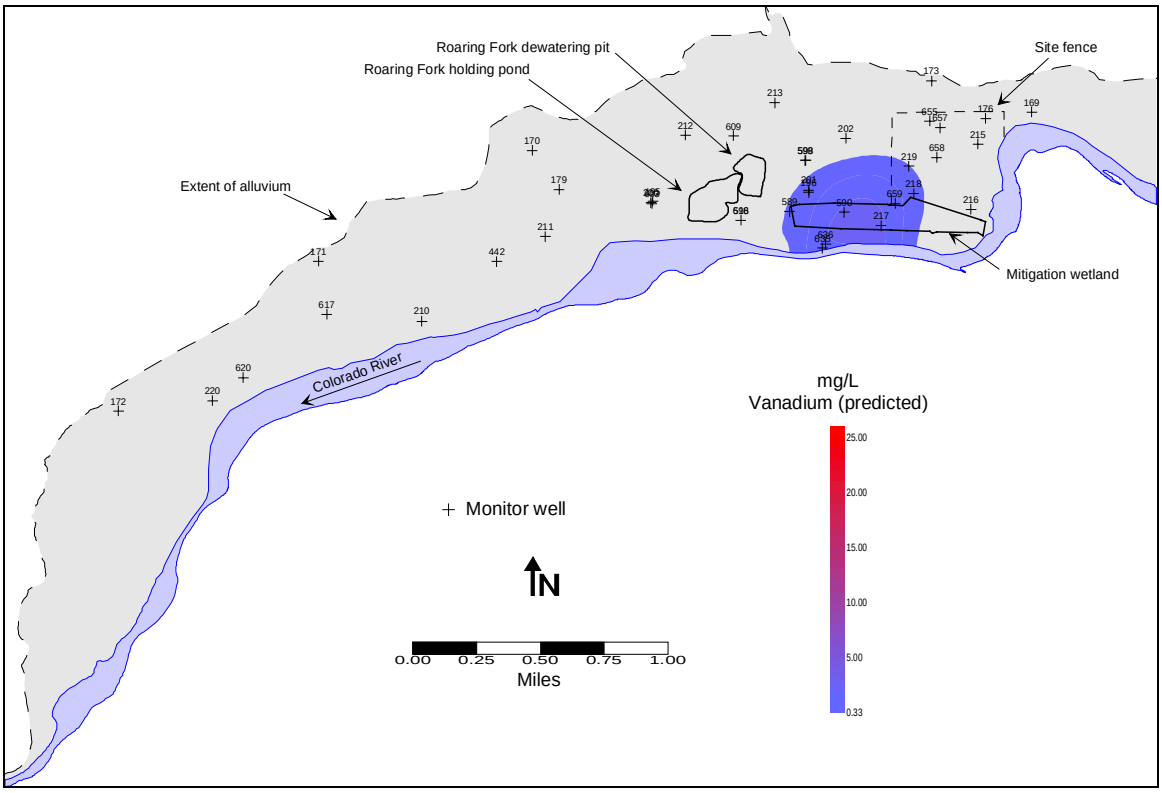

Figure 5–55. Predicted Vanadium Concentrations in the Ground Water at the New Rifle Site at 100 Years Based on the January 1999 Sampling Data—100 Simulations

5.4 Ecology

Field investigations were conducted in 1998 (Section 4.9) to complete the assessment of ecological risks associated with site-related contaminated ground water and to assist in developing the site conceptual model required for simulating natural flushing. Results of the ecological risk assessment are presented in Section 6.0. Plant ecology as it relates to the development of the site conceptual model is presented below.

5.4.1 Plant Evapotranspiration

Vegetation can influence both recharge and discharge from the alluvial aquifer depending on the health of the plant communities, soil properties, and depth to ground water. Vegetation at the site and surrounding areas were mapped to support the water balance model developed as part of the numerical flow and transport model (Appendix D). Several types of vegetation were identified as shown in Figure 5–56.

5.4.2 Methods

Most of the site and an equal area immediately downgradient of the west fence boundary are dominated by upland grassland as a result of the revegetation at the conclusion of the surface remediation project. Phyreatophyte populations of greasewood and cottonwood dominate the landscape west of the Roaring Fork gravel operations. At this western extent of the project area the greasewood occurs mostly north of the railroad tracks while cottonwood is the most prevalent vegetation south of the railroad tracks.

Three general areas having similar ET characteristics are shown in Figure 5–57. These ET areas were delineated based on vegetation, soil type, and historical and current land use. Literature values for upland grassland range from 3 to 35 centimeters per year (cm/yr) (Nichols 1993; Branson et al. 1981). ET can remove almost all precipitation entering deep, fine- to medium-textured soils in upland semiarid areas where a diverse, late-succession plant community dominates (Richard et al. 1988). Where these conditions occur, recharge is negligible. In contrast, at the New Rifle site, recharge may be as high as 50 percent of annual precipitation because of the coarse-textured soils and grass monoculture. Coarse-textured soils store little water (Hillel 1982) and ET may be considerably less from grass monocultures than from native shrub-grass vegetation (Waugh et al. 1994, Gee et al. 1991). If, in the future, the plant community at the New Rifle site trends toward phreatophytic shrubs such as greasewood, then the site may shift from a recharge zone to a discharge zone.

Transpiration rates for greasewood populations range from 20 to 160 cm/yr (Nichols 1993; Branson et al. 1981). For the cottonwood and tamarisk riparian vegetation the transpiration rate ranges from 30 to 260 cm/yr. These downgradient areas are discharge zones.

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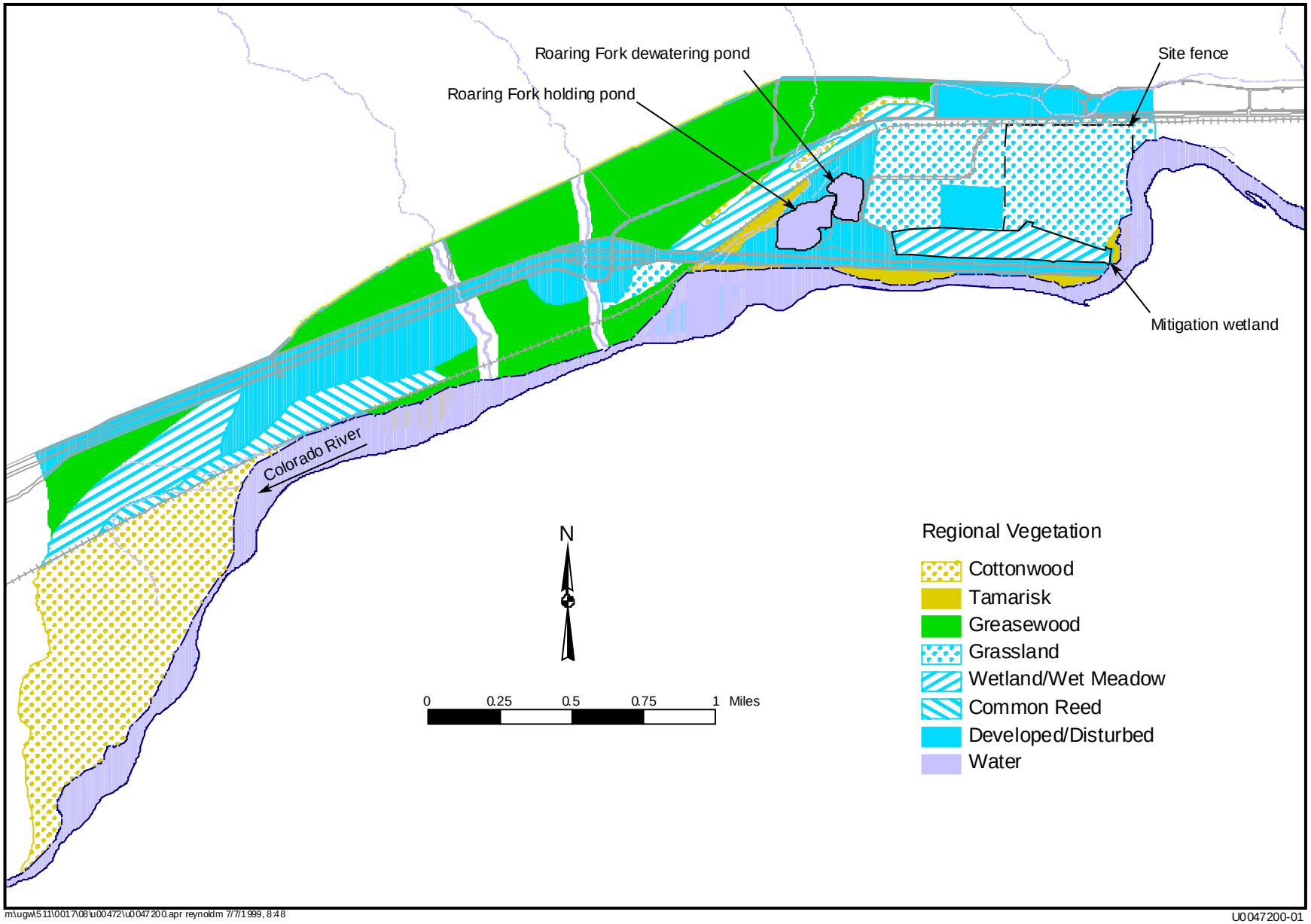

Figure 5–56. Regional Vegetation at the New Rifle Site

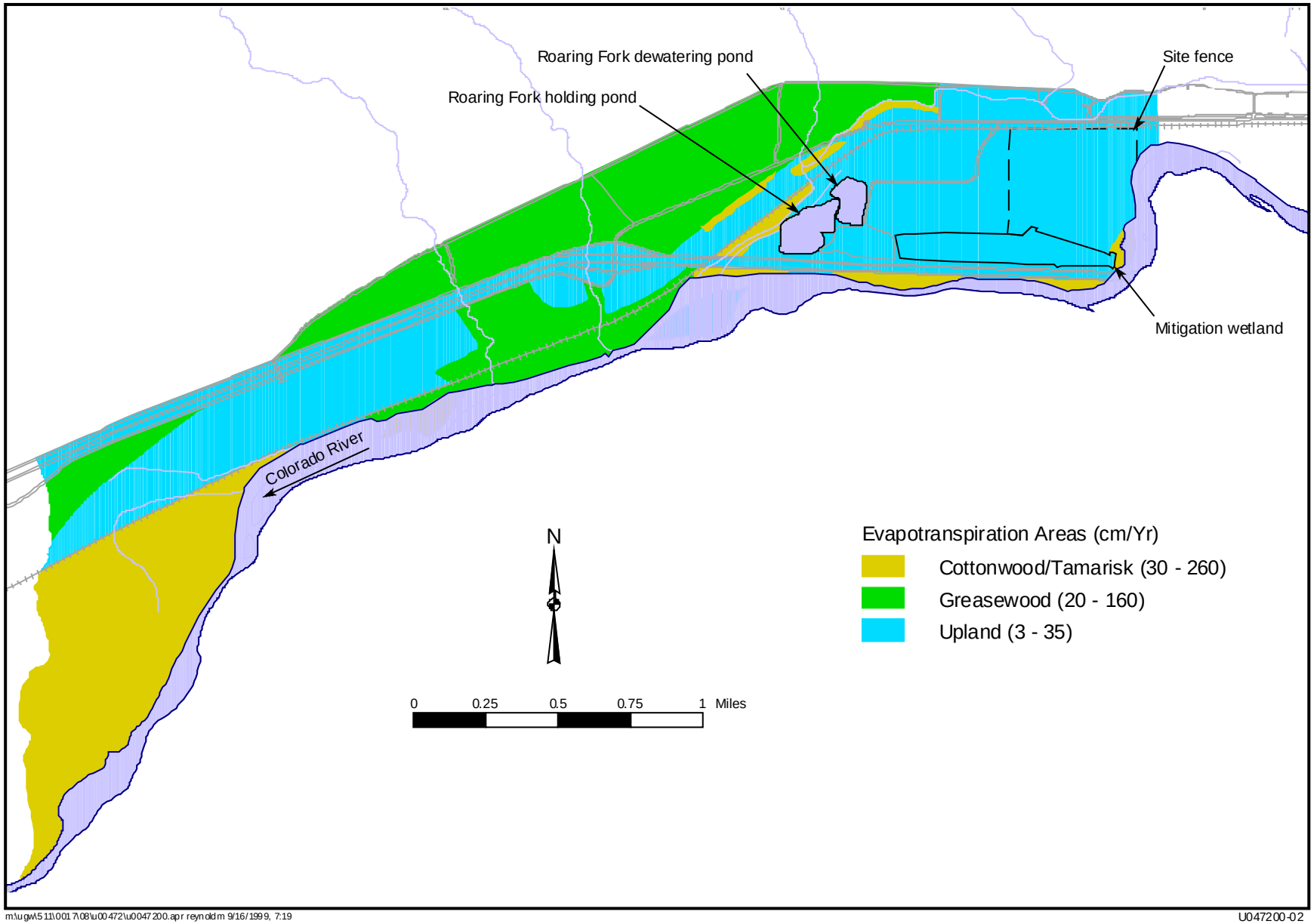

Figure 5–57. Evapotranspiration Areas at the New Rifle Site

6.0 Summary of Human Health and Ecological Risk

6.1 Human Health Risk Assessment

A BLRA was previously prepared for the New Rifle site (DOE 1996a) according to methods provided in the PEIS (DOE 1996b). Much of the data used in that risk analysis were collected before completion of surface remediation in 1996 (data for characterizing the contaminant plume were collected from 1990 to 1994). Additional wells have been installed in both the alluvial aquifer and Wasatch Formation in 1998 and more recent samples have been collected (see Section 4.0). Several contaminants have shown significant changes, often decreases, in concentration since completion of the original BLRA. This necessitates a reevaluation of COPC identification and assessment of associated risks. The intent of this BLRA update is to use those earlier results and conclusions as a starting point from which to evaluate the more recent data.

6.1.1 Summary of 1996 Risk Assessment Methodology and Results

The 1996 BLRA identified 24 contaminants associated with the alluvial aquifer at the New Rifle site as being at levels statistically above background concentrations for the area. This initial list of contaminants was screened to eliminate contaminants of low toxicity and high dietary ranges. These two steps eliminated four contaminants each, resulting in the following COPC list: ammonium, antimony, arsenic, cadmium, chloride, fluoride, iron, lead, manganese, molybdenum, nitrate, selenium, sodium, sulfate, uranium, and vanadium. The radionuclides lead-210, polonium-210, radium-226, and thorium-230 were all retained as COPCs as well because they are known human carcinogens. Further risk analysis was performed on all these contaminants.

A number of potential exposure routes were considered for alluvial ground water: ingestion of ground water as drinking water in a residential setting, dermal contact while bathing, ingestion of ground water-irrigated produce, ingestion of milk or meat from ground water-fed livestock, and ingestion of fish from the Colorado River. Only the ground water ingestion pathway was retained for quantitative analysis. The others were either considered to represent negligible risk or data were unavailable to evaluate them in any meaningful way. Consequently, the use of ground water as drinking water in a residential setting was evaluated probabilistically. For additional information on other potential exposure routes and the probabilistic methodology, see the BLRA (DOE 1996a).

Results of the BLRA showed that the noncarcinogenic contaminants that could result in mild to severe adverse health effects were ammonium, arsenic, manganese, nitrate, selenium, sodium, sulfate, chloride, molybdenum, fluoride, iron, and vanadium. Cadmium, antimony, and lead were shown to be of little concern. Noncarcinogenic risks from uranium exposure were questionable due to lack of reliable toxicological data. For carcinogens, the total individual excess lifetime cancer risk from exposure to all radionuclides combined exceeded EPA's upper bound acceptable risk level of 1E-04. The major radiological contributor to carcinogenic risk was uranium; risks posed by each of the other carcinogens individually are within EPA's acceptable risk range (1E-04 to 1E-06). Additionally, the total individual excess lifetime cancer risk for arsenic was also above the upper bound acceptable risk level. (For further discussion of toxicological effects from exposure to specific contaminants, see DOE 1996a).

Risks associated with use of ground water from the Wasatch Formation were not specifically addressed in the original BLRA. At that time, concentrations of contaminants in the Wasatch were lower than those in the alluvial aquifer. However, it was assumed that because the Wasatch and alluvial aquifer are hydraulically connected, eventually contaminant levels in the Wasatch could reach those of the alluvial aquifer. Based on this worst-case assumption, risks calculated for the alluvial aquifer also represent maximum risks that could result from future use of contaminated Wasatch ground water.

6.1.2 BLRA Update

6.1.2.1 Alluvial Aquifer COPC List Update

This BLRA uses the COPC list from the original BLRA as a starting point to evaluate current data. Table 6–1 lists the COPCs identified in the 1996 BLRA along with a summary of historical plume data (from the 1996 SOWP) and current (1998–99) plume and background data. MCLs and RBCs; (EPA 1998b) are also included for comparison of data to benchmarks. Background locations are those used for Old Rifle plus two additional locations upgradient from New Rifle and outside the influence of either site. Historic plume data include on-site and immediately downgradient wells. Current plume data are for on-site wells only.

The RBCs presented in Table 6–1 for a given contaminant represents a concentration in drinking water that would be protective of human health provided that:

- A residential exposure scenario with standard default assumptions (EPA 1989a) is appropriate.
- Ingestion of contaminated drinking water is the only exposure pathway that applies. (Note: Does not apply to ammonia. See discussion in Section 6.1.2.2.)
- The contaminant contributes nearly all of the health risk (i.e., there are no additive risks from the presence of multiple contaminants).
- EPA's lower bound risk level of 1×10^{-6} for carcinogens and a HI of 1 for noncarcinogens is appropriate.

If any of these assumptions is *not* true, contaminant levels at or below RBCs cannot automatically be assumed to be protective. For example, if multiple contaminants are present in drinking water, a single contaminant may be below its RBC but still be a significant contributor to the total risk posed by drinking the water. However, if an RBC is exceeded, it is an indication that further evaluation of the contaminant is warranted. RBCs are intended to be used in screening-level evaluations. For additional explanation on the derivation of RBCs, refer to EPA 1998b. Note that the RBC for vanadium is based on data from vanadium pentoxide. The draft SOWP (DOE 1996c) indicated that vanadium at the New Rifle site occurs in this form.

A comparison of historical and current plume data indicates that concentrations of many constituents have decreased over time. This decrease is particularly striking for ammonia, arsenic, chloride, nitrate, sodium, and sulfate. Chloride and sodium concentrations in on-site wells have decreased to within the range of background concentrations and can therefore be eliminated from further consideration as COPCs. Lead-210, polonium-210, and thorium-230

Table 6–1. New Rifle Site—Data Summary

Contaminant	No.	Minimum mg/L	Maximum mg/L	Mean mg/L	MCL mg/L	RBC mg/L
Ammonia (as NH_4)					N/A	0.28 as NH_3
Background	16/19	0.001	0.61	0.1358		193 as NH_4 (back-calculated using site pH, T)
Current Plume	17/18	0.0038	475	145.5		
Historic Plume	19/19	506	1745	1030		
Arsenic					0.05	0.011N
Background	8/19	0.00031	0.0034	N/A		0.000045C
Current Plume	15/18	0.0001	0.304	0.0391		
Historic Plume	2/2	0.97	1.3	1.1		
Chloride					N/A	N/A
Background	19/19	66	274	158.8		
Current Plume	18/18	27.1	177	112.8		
Historic Plume	33/33	55	1750	420		
Fluoride					4	2.2N
Background	19/19	0.422	1.45	0.8409		
Current Plume	18/18	0.477	5.5	2.505		
Historic Plume	4/4	6.2	8.1	8.1		
Iron					N/A	11N
Background	11/19	0.003	3.03	0.3784		
Current Plume	10/18	0.003	14.3	1.592		
Historic Plume	2/2	16	17	17		
Manganese					N/A	1.7N
Background	19/19	0.0064	3.66	0.8599		
Current Plume	18/18	0.34	4.55	2.103		
Historic Plume	20/20	9	13	9.9		
Molybdenum					0.1	0.18N
Background	19/19	0.0034	0.0345	0.0189		
Current Plume	18/18	0.0101	6.84	2.217		
Historic Plume	23/23	2.3	3.7	2.9		
Nitrate					44	255N (as Nitrate)
Background	15/19	0.011	20.6	4.025		
Current Plume	15/18	0.011	324	35.14		
Historic Plume	7/7	552	1110	784		
Selenium					0.01	0.18N
Background	13/19	0.001	0.0298	0.0109		
Current Plume	17/18	0.00066	0.782	0.09311		
Historic Plume	14/34	0.002	0.3	N/A		
Sodium					N/A	N/A
Background	19/19	164	655	424.9		
Current Plume	18/18	75.1	648	314.3		
Historic Plume	22/22	1300	3850	2390		
Sulfate					N/A	N/A
Background	19/19	482	2240	1101		
Current Plume	18/18	121	2840	1298		
Historic Plume	21/21	1560	12600	7350		
Uranium					0.044	0.11N
Background	19/19	0.0055	0.0594	0.0372		
Current Plume	18/18	0.0103	0.395	0.1103		
Historic Plume	6/6	0.24	0.37	0.29		
Vanadium					N/A	0.33
Background	5/19	0.001	0.0023	N/A		
Current Plume	15/18	0.001	25.3	3.168		
Historic Plume	12/12	0.59	2.8	1.3		
Lead-210 (pCi/L)					N/A	0.068C
Background	0/19	0.9	1.33	N/A		
Current Plume	1/18	0.93	1.24	N/A		
Historic Plume	N/A	0	4.6	0.7		

Contaminant	No.	Minimum mg/L	Maximum mg/L	Mean mg/L	MCL mg/L	RBC mg/L
Polonium-210 (pCi/L)					N/A	0.14C
Background	1/19	0.05	0.18	N/A		
Current Plume	2/18	0.04	2.75	N/A		
Historic Plume	N/A	0	1.6	0.3		
Thorium-230 (pCi/L)					N/A	1.22C
Background	0/19	0.8	2.3	N/A		
Current Plume	0/18	0.8	2	N/A		
Historic Plume	N/A	0	53	0.3		

Alluvial background wells: RFO–291, –292, –597, –598, –606, –658, –659, –660; and RFN–169, –173
Current plume wells: 301–310, 654, 655–659, 662, 663 (all on–site)
Historic plume wells: RFN–581, –583, –584, –587, –589, –590, –594, –595, –599, –618, –619 (data from DOE 1996a)
RBC = risk–based concentration (EPA 1998a)
C = carcinogenic risk
N = noncarcinogenic risk

For mean calculations, values for samples below detection set at one-half the detection limit

have a very low frequency of detection. These decay products of uranium can also be eliminated as COPCs, particularly if uranium is retained for further evaluation (as is recommended here).

Sulfate concentrations on-site are only marginally higher than those associated with background. No MCL or risk-based levels have currently been established for sulfate in drinking water. The secondary drinking water standard for sulfate is 250 mg/L, based on considerations of taste and smell. Both on-site and background ground water greatly exceed this unenforceable standard. A recent report by EPA (EPA 1999) indicates that levels of sulfate in drinking water up to 1,500 mg/L may result in no adverse health effects. The means of both background and on-site ground water are below this concentration. Based on the lack of toxicity data, and the fact that on-site ground water is only marginally above background, sulfate can also be eliminated from further consideration as a COPC. The remaining constituents listed in Table 6–1 will be retained for evaluation in this BLRA update.

6.1.2.2 Risk Assessment Methodology

As mentioned previously, the original BLRA considered several potential routes of exposure to contaminants and eliminated all but one, ingestion of ground water in a residential setting, as insignificant. Therefore, the ground water ingestion pathway is the only route of exposure considered in this BLRA update (with the exception of ammonia as discussed below). Note that all risks discussed in this document are hypothetical with respect to human health. On-site ground water is currently not being used for any purpose and institutional controls will be put in place to restrict its use in the future. Private water wells are in use downgradient of the site; however, all water is treated to safe levels prior to use. It is possible that additional private wells could be installed into the contaminated alluvial aquifer in the future. Treatment systems will likely be required in conjunction with any future well installation that takes place. Efforts are currently underway to identify appropriate well installation restrictions and requirements and to determine how these measures can be effectively enforced. Risks discussed in this BLRA update pertain only to use of untreated ground water.

Risk calculations presented here follow EPA's *Risk Assessment Guidance for Superfund Methodology* (EPA 1989b), which involves determining a point estimate for excess cancer risk from current or potential carcinogenic exposures and a hazard quotient (HQ; ratio of exposure

intake to an acceptable intake) for noncarcinogenic exposures. It is assumed that the receptors for ground water at the New Rifle site are residents who use untreated alluvial ground water as their primary source of drinking water. As noted above, this scenario is unlikely because of institutional controls and restrictions that will probably be put in place; however, it is consistent with the scenario evaluated in the original BLRA.

The original BLRA calculated noncarcinogenic risks using a probabilistic approach. Essentially, this means that instead of using a single value for each parameter required in the risk calculations (e.g., ground water concentration, body weight, frequency of exposure), a range of values with a given probability distribution was used. By performing numerous iterations of the standard risk calculations, with a value selected at random from each parameter distribution, a range of exposures, and associated risk results. The original BLRA reported results for the most sensitive receptor population modeled—children.

In this update, which uses point-exposure doses, single values are used for each parameter required in the risk calculations. Calculations to determine contaminant intakes use standard exposure factors for the adult population (EPA 1989a). Ground water concentrations used to calculate risks associated with the ingestion of contaminated ground water are derived from the most recent rounds of sampling (1998 and 1999). A separate set of calculations was performed for on-site wells and downgradient wells to provide information required for selecting appropriate monitoring and compliance strategies. To get a reasonable worst-case risk estimate for on-site wells, the 95 percent upper confidence limit (UCL_{95}) for the means of contaminant concentrations was used in the risk. The UCL_{95} was used instead of the maximum contaminant concentrations because in many cases the maximum concentrations were represented by results of one or two analyses that were an order of magnitude or more higher than the rest of the on-site samples. The mean concentrations were also used to determine average on-site risks. For downgradient wells, risks were calculated using both the maximum and mean contaminant concentrations.

Compared with the original, probabilistic BLRA, use of adult exposure data is probably less conservative than use of the exposure data for children. However, using UCL_{95} and maximum contaminant concentrations and point-exposure dose calculations is probably more conservative; the net effect is to produce comparably conservative results. For purposes of making risk management decisions, results of both methodologies are usable and each has its advantages and limitations.

In this update, risks associated with ammonia were generally calculated as described above for other noncarcinogens with one important exception. For all contaminants except ammonia, risks were determined for ingestion of contaminated ground water (i.e., an oral exposure route) in a residential setting. The major risks resulting from ammonia exposure are not through oral ingestion in ground water, but rather from inhalation of ammonia in the gaseous form through volatilization from ground water. Risks were calculated using default inhalation exposure parameters for a residential setting (EPA 1991). Analytical results for ammonia were reported as NH_4. The actual amount of ammonia gas, NH_3, available for volatilization was calculated for site-specific temperature and pH (using data compiled by Emerson et al. 1975).

Note that risks associated with ammonia for a residential setting requires that exposure occurs within a closed structure (i.e., a residence) in which volatilized ammonia is trapped through use of ground water for all domestic purposes (drinking, bathing, laundry, etc.). For exposure

scenarios where a closed structure is absent (e.g., irrigation and agricultural), volatilized ammonia would quickly dissipate to the atmosphere and risks would be negligible (at least at the concentrations presented in ground water at the New Rifle site). Therefore, exposure to ammonia is only of concern in a residential setting.

The same methodology was used to calculate carcinogenic risks for this BLRA update as was used in the original BLRA (i.e., receptors are adults with exposure averaged over 70 years). For all risk calculations, benchmarks for acceptable contaminant intakes (e.g., reference doses, slope factors) are the best available data from standard EPA sources (e.g., Integrated Risk Information System, Region III Risk-Based Concentration Table).

6.1.2.3 Results—Alluvial Aquifer

Results of the risk calculations are included in Tables 6–2 and 6–3 for on-site and downgradient locations, respectively. Standard risk equations and default parameter values are also provided. For noncarcinogens, the tables also show the percentage that each contaminant contributes to the total HI (or overall risk).

The following major observations are based on these results:

- The greatest potential noncarcinogenic risks are associated with the use of on-site ground water. For both UCL_{95} and mean concentrations, the major risk contributors are molybdenum and vanadium and, to a lesser degree, arsenic. Individual risks associated with most other contaminants are only marginally above acceptable levels. Risks associated with iron are at acceptable levels and are negligible with respect to total risks posed by on-site ground water contamination.

- While noncarcinogenic risks associated with use of downgradient ground water are lower, on the whole, than those associated with on-site ground water, risks from nitrate are significantly higher. For maximum downgradient concentrations, the major risk contributors are molybdenum, nitrate, and manganese and, to a lesser extent, uranium and vanadium. Risks associated with mean vanadium concentrations are negligible. All noncarcinogenic risks associated with arsenic, iron, and selenium are below maximum acceptable levels.

- Carcinogenic risks posed by arsenic exceed EPA's maximum acceptable risk of 1 E-4 for both on-site and downgradient ground water. However, it should be noted that acceptable risk-based concentrations of arsenic are below the detection limit for this analyte. Therefore, all calculated carcinogenic risks will exceed acceptable levels even if the detection limit is used as a conservative estimate for nondetectable concentrations.

- All carcinogenic risks posed by uranium (total combined U234 and U238) are within or below the acceptable risk range for on-site and downgradient locations. However, only one on-site and five downgradient wells were analyzed for these isotopes, so limited data were available for evaluation.

Based on the negligible risks posed by iron in on-site and downgradient alluvial wells, it can be eliminated from further consideration as a COPC. Concentrations of arsenic, molybdenum, nitrate, selenium, and uranium all exceed UMTRA drinking water standards on-site and must be retained as COPCs for purposes of development of an appropriate compliance strategy.

Table 6–2. New Rifle Risk Calculations
1998-1999 On-Site Alluvial Aquifer Data

$$\text{Intake} = \frac{CW \times IR \times EF \times ED}{BW \times AT}$$

where[a]:

Intake is in (mg/kg-d)
CW = chemical concentration in water (mg/L); site-specific
IR = ingestion rate (L/d); 2 L/day adult; default
ED = exposure duration (years); 30 yrs for adult; default
EF = exposure frequency (d/yr); 350 days/yr; default
BW = body weight (kg); 70 kg adult; default
AT = averaging time; ED x 365 d/yr non-carc., 70yr x 365 d/yr carc.

Hazard Quotient (HQ) = Intake/Reference Dose (RfD)
Risk = Chronic Daily Intake (averaged over 70 years) x Slope Factor
For radionuclides, Risk = SF x CW x IR x EF x ED (slope factor accounts for average lifetime risk); concentrations expressed in pCi/L

Contaminant		CW-95%	IR	EF	ED	BW	AT	Intake-95%	RfD[b]	HQ-95%	%Risk	CW-MEAN	Intake-mean	HQ-MEAN	%Risk
		mg/L										mg/L			
Arsenic		0.0748	2	350	30	70	10950	0.0020	0.0003	6.831	13.20	0.0391	0.001071	3.5708	12.09
Fluoride		3.5138	2	350	30	70	10950	0.0963	0.06	1.604	3.10	2.505	0.068630	1.1438	3.87
Iron		3.3976	2	350	30	70	10950	0.0931	0.3	0.310	0.60	1.592	0.043616	0.1454	0.49
Manganese		2.8194	2	350	30	70	10950	0.0772	0.047	1.643	3.18	2.103	0.057616	1.2259	4.15
Molybdenum		3.5191	2	350	30	70	10950	0.0964	0.005	19.283	37.27	2.217	0.060740	12.1479	41.13
Nitrate		75.9516	2	350	30	70	10950	2.0809	7	0.297	0.57	35.14	0.962740	0.1375	0.47
Selenium		0.1851	2	350	30	70	10950	0.0051	0.005	1.014	1.96	0.0931	0.002551	0.5101	1.73
Uranium		0.1598	2	350	30	70	10950	0.0044	0.003	1.459	2.82	0.1103	0.003022	1.0073	3.41
Vanadium		6.3385	2	350	30	70	10950	0.1737	0.009	19.295	37.29	3.168	0.086795	9.6438	32.65
									HI =	51.738			HI=	29.5326	
Non-carcinogens - Inhalation through water use in residential setting*															
Ammonia (max)		0.689	15	350	30	70	10950	0.1416	0.0286	4.950					
Ammonia (mean)		0.226	15	350	30	70	10950	0.0464	0.0286	1.624					
*IR = 15 m^3/d of air default; concentration in air = water concentration x water-to-air volatilization factor x conversion factor															
Default volatilization factor = 0.0005; conversion factor is 1,000L/m^3															
Maximum NH_3 in on-site New Rifle ground water is 1.38 mg/L, mean on-site is 0.452 mg/L															
Carcinogens - Groundwater Ingestion Only (Adults)															
Contaminant		CW	IR	EF	ED	BW	AT	Intake	SF[b]	Risk					
Arsenic	95%	0.0748	2	350	30	70	25550	0.0008783	1.5	1.32E-03					
	mean	0.0391	2	350	30	70	25550	0.0004591	1.5	6.89E-04					
U234+238	1 sample	9.1	2	350	30	na	na	1.91E+05	4.36E-11	8.33E-06					

[a] All exposure factors are from EPA 1989b

[b] Data are mainly from EPA's Integrated Risk Information System (IRIS) or Health Effects Assessment Summary Tables (HEAST); other values are from EPA Region III Risk-Based Concentration Table

Table 6–3. New Rifle Risk Calculations
1998-1999 Downgradient Alluvial Aquifer Data

$$\text{Intake} = \frac{CW \times IR \times EF \times ED}{BW \times AT}$$

where[a]:

Intake is in (mg/kg-d)
CW = chemical concentration in water (mg/L); site-specific
IR = ingestion rate (L/d); 2 L/day adult; default
ED = exposure duration (years); 30 yrs for adult; default
EF = exposure frequency (d/yr); 350 days/yr; default
BW = body weight (kg); 70 kg adult; default
AT = averaging time; ED x 365 d/yr non-carc., 70yr x 365 d/yr carc.

Hazard Quotient (HQ) = Intake/Reference Dose (RfD)
Risk = Chronic Daily Intake (averaged over 70 years) x Slope Factor
For radionuclides, Risk = SF x CW x IR x EF x ED (slope factor accounts for average lifetime risk); concentrations expressed in pCi/L

Contaminant		CW-max	IR	EF	ED	BW	AT	Intake-max	RfD[b]	HQ-max	%Risk	CW-mean	Intake-mean	HQ-mean	%Risk
		mg/L										mg/L			
Arsenic		0.008	2	350	30	70	10950	0.0002	0.0003	0.731	2.12	0.001	0.000027	0.0913	1.28
Fluoride		6.39	2	350	30	70	10950	0.1751	0.06	2.918	8.46	1.447	0.039644	0.6607	9.27
Iron		4.48	2	350	30	70	10950	0.1227	0.3	0.409	1.19	0.003	0.000082	0.0003	0.00
Manganese		8.82	2	350	30	70	10950	0.2416	0.047	5.141	14.90	2.229	0.061068	1.2993	18.23
Molybdenum		3.15	2	350	30	70	10950	0.0863	0.005	17.260	50.02	0.6296	0.017249	3.4499	48.40
Nitrate		713	2	350	30	70	10950	19.5342	7	2.791	8.09	187.3	5.131507	0.7331	10.28
Selenium		0.0274	2	350	30	70	10950	0.0008	0.005	0.150	0.44	0.0055	0.000151	0.0301	0.42
Uranium		0.209	2	350	30	70	10950	0.0057	0.003	1.909	5.53	0.0939	0.002573	0.8575	12.03
Vanadium		1.05	2	350	30	70	10950	0.0288	0.009	3.196	9.26	0.002	0.000055	0.0061	0.09
									HI =	34.505			HI=	7.1284	
Non-carcinogens - Inhalation through water use in residential setting*															
Ammonia (max)		0.97	15	350	30	70	10950	0.1993	0.0286	6.969					
Ammonia (mean)		0.221	15	350	30	70	10950	0.0454	0.0286	1.588					
*IR = 15 m^3/d of air default; concentration in air = water concentration x water-to-air volatilization factor x conversion factor															
Default volatilization factor =0.0005; conversion factor is 1,000L/m^3															
Maximum NH_3 in New Rifle ground water is 1.9 mg/L, mean downgradient is 0.442 mg/L															
Carcinogens - Groundwater Ingestion Only (Adults)															
Contaminant		**CW**	**IR**	**EF**	**ED**	**BW**	**AT**	**Intake**	**SF[b]**	**Risk**					
Arsenic	max	0.008	2	350	30	70	25550	9.39335E-05	1.5	1.41E-04					
	mean	0.001	2	350	30	70	25550	1.17417E-05	1.5	1.76E-05					
U234+238	max	71.9	2	350	30	na	na	1.51E+06	4.36E-11	6.58E-05					
	mean	3.79E+01	2	350	30	na	na	7.97E+05	4.36E-11	3.47E-05					

[a] All exposure factors are from EPA 1989b
[b] Data are mainly from EPA's Integrated Risk Information System (IRIS) or Health Effects Assessment Summary Tables (HEAST); other values are from EPA Region III Risk-Based Concentration Table

Ammonia, fluoride, manganese, and vanadium all exceed risk-based concentrations for on-site locations. Fluoride concentrations exceed the current MCL (though this standard is undergoing review). No MCL exists for manganese, but on-site concentrations also exceed the secondary drinking water standard. Downgradient of the site, mean concentrations of molybdenum, nitrate, and uranium exceed UMTRA ground water standards. Arsenic is below the UMTRA standard in all downgradient wells. Ammonia, selenium, vanadium, and fluoride exceed standards or risk-based concentrations only in wells immediately downgradient of the New Rifle site and within the bounds of the Roaring Fork gravel pit operation. Downgradient of the gravel operation, these four contaminants decrease to below standards or RBC. Mean downgradient concentrations of manganese exceed RBCs and secondary drinking water standards and elevated concentrations extend beyond the boundary of the gravel mining operation.

Based on the above discussion, it can be concluded the greatest potential risks posed by site-related contamination are attributable to noncarcinogenic contaminants. Most pervasive are manganese, molybdenum, nitrate, and uranium. Elevated risks associated with ammonia, arsenic, fluoride, selenium, and vanadium are of concern for only on-site and immediately downgradient locations. Concentrations of these constituents decrease to acceptable levels west of the Roaring Fork gravel operation. Iron concentrations pose negligible risks in both on-site and downgradient alluvial ground water.

6.1.2.4 Wasatch Formation

As in the 1996 BLRA, risks specifically associated with use of ground water from the Wasatch Formation are not determined for this BLRA update. However, concentrations of all COPCs in ground water from on-site and downgradient Wasatch wells are lower than those from respective alluvial wells. Therefore, as concluded in the original BLRA, risks determined for the alluvial aquifer represent future worst-case estimates for the Wasatch Formation.

6.2 Ecological Risk Assessment

An Ecological Risk Assessment (ERA) was previously prepared for the New Rifle site as part of the BLRA (DOE 1996a). That qualitative, screening-level assessment was based on limited data that were collected before the completion of surface remediation. The additional data collected in 1998 reflects post-remediation site conditions and permits a more quantitative assessment of ecological risk. This section summarizes the initial 1996 ERA and briefly describes the methodology and results of the ERA update. The approach used for the ERA update was based on guidance in *Guidelines for Ecological Risk Assessment* (EPA 1998a) and *Framework for Ecological Risk Assessment* (EPA 1992). Details on the quantitative ecological risk assessment process are provided in Calculation Set U0060400; Appendix E contains all data and equations used to estimate quantitative risks to ecological receptors.

6.2.1 Summary of 1996 Ecological Risk Assessment

The 1996 screening-level ERA identified potential pathways of exposure for ecological receptors, identified COPCs for each pathway, and compared contaminant concentrations in those media to appropriate ecological benchmarks. The exposure scenarios evaluated included:

- Direct contact of terrestrial plants with contaminated ground water and uptake through root systems. It was assumed that root systems of plants could extend below the shallow water table at the site and contact contamination.

- Direct contact of plants and wildlife with ground water discharging into the mitigation wetland (referred to here as the New Rifle wetland located on the downgradient edge of the New Rifle site).

- Direct contact of plants and aquatic and terrestrial wildlife with surface water and sediment contained in the Roaring Fork holding pond (Roaring Fork pond) and the Colorado River.

Surface water, sediment, and ground water samples were collected from the site vicinity for comparison with background samples to identify COPCs and assess them against screening benchmarks. Surface water data from the Colorado River near the site indicated ammonium and iron exceeded background concentrations but did not exceed state standards for the protection of aquatic life in surface water or screening benchmarks for terrestrial wildlife and plants. Surface water quality in the Roaring Fork pond was affected by site activities, as evidenced by the elevated concentrations of several COPCs. Nonetheless, no COPCs were detected at concentrations higher than state water quality standards for the protection of aquatic life or terrestrial plant and wildlife screening benchmark values. Nitrate levels in the pond exceeded levels protective of livestock watering; however, the pond was (and still is) protected from livestock access.

In general, sediment concentrations for the COPCs detected at the Colorado River locations were similar to background concentrations, suggesting activities at the site did not measurably affect sediment quality in the river.

At the New Rifle wetland, several contaminants were identified that could represent a potential hazard to ecological receptors. Cadmium, chloride, iron, manganese, and selenium in ground water collected near the wetland site exceeded their respective federal or state water quality standards for the protection of aquatic life via short-term and chronic exposure. Ground water concentrations exceeded two contaminant-specific screening benchmarks (arsenic and vanadium) for terrestrial wildlife and seven benchmarks (arsenic, cadmium, iron, manganese, molybdenum, vanadium, and zinc) for terrestrial plants. Tissue from biota was not analyzed, but bioaccumulation of selenium, cadmium, and manganese was considered possible in the mitigation wetlands. Additional sampling and monitoring were recommended to further refine the results of this screening-level assessment. A complete list of the tentatively identified COPCs and their respective media of concern are presented in Table 6–4. These results were used as the starting point for additional data collection and update of the ERA.

Table 6–4. Summary of Ecological Contaminant of Potential Concern in Ground Water, Surface Water, and Sediments

Constituents Above Background[a]	New Rifle Ground Water[b]	Colorado River Surface Water	Colorado River and Pond Sediment
Ammonium	X	X	
Arsenic	X		X
Cadmium	X		X
Calcium	X		
Chloride	X		
Fluoride	X		
Iron	X	X	X
Manganese	X		X
Molybdenum	X		X
Nitrate	X		
Phosphate	X		
Potassium	X		
Selenium	X		X
Silica	X		
Sodium	X		
Strontium	X		
Sulfate	X		
Uranium	X		X
Vanadium	X		X
Zinc	X		X

[a]Greater than concentrations in a reference area (e.g., upgradient well, upriver surface water, or upgradient pond) at the 90 percent confidence level.
[b]Because the alluvial aquifer discharges to the mitigation wetland, alluvial aquifer water quality data were used to evaluate potential adverse effects to wetland ecology.

6.2.2 ERA Update

Figure 6–1 shows the general phases and activities involved in completing the New Rifle ERA model in the context of EPA guidance (EPA 1998a). The preliminary problem formulation phase was completed largely through the initial BLRA (DOE 1996a). The 1998 field investigation activities were designed to fill data gaps and further refine the initial site conceptual model. Data also were collected to support exposure analysis and risk estimation through development of a quantitative risk assessment (QRA). Sections 6.2.2.2 and 6.2.2.3 briefly explain the methodology and results of the exposure analysis and risk estimation process. Appendix E describes the exposure analysis in more detail and contains the data used in calculating exposure intakes for receptors. Exposure intakes are used in the risk estimation phase to determine numerical risk estimates that represent potential risks to a receptor or receptor population. The risk estimates are based on comparing site-specific exposure intakes with intakes from other studies that have been shown to produce a certain response in a given receptor. The site-specific risk estimates are largely theoretical and based on statistical tests and calculations rather than direct observation.

In parallel to determining numerical risk estimates, a plant uptake study was designed and implemented to determine actual observable ecological effects of site-related contamination on wetlands plants. Results of this study will give an indication of actual ecological effects that can be expected during future revegetation of the New Rifle wetland. Results will not only be used for assessing ecological effects of current site conditions, but also for planning reconstruction

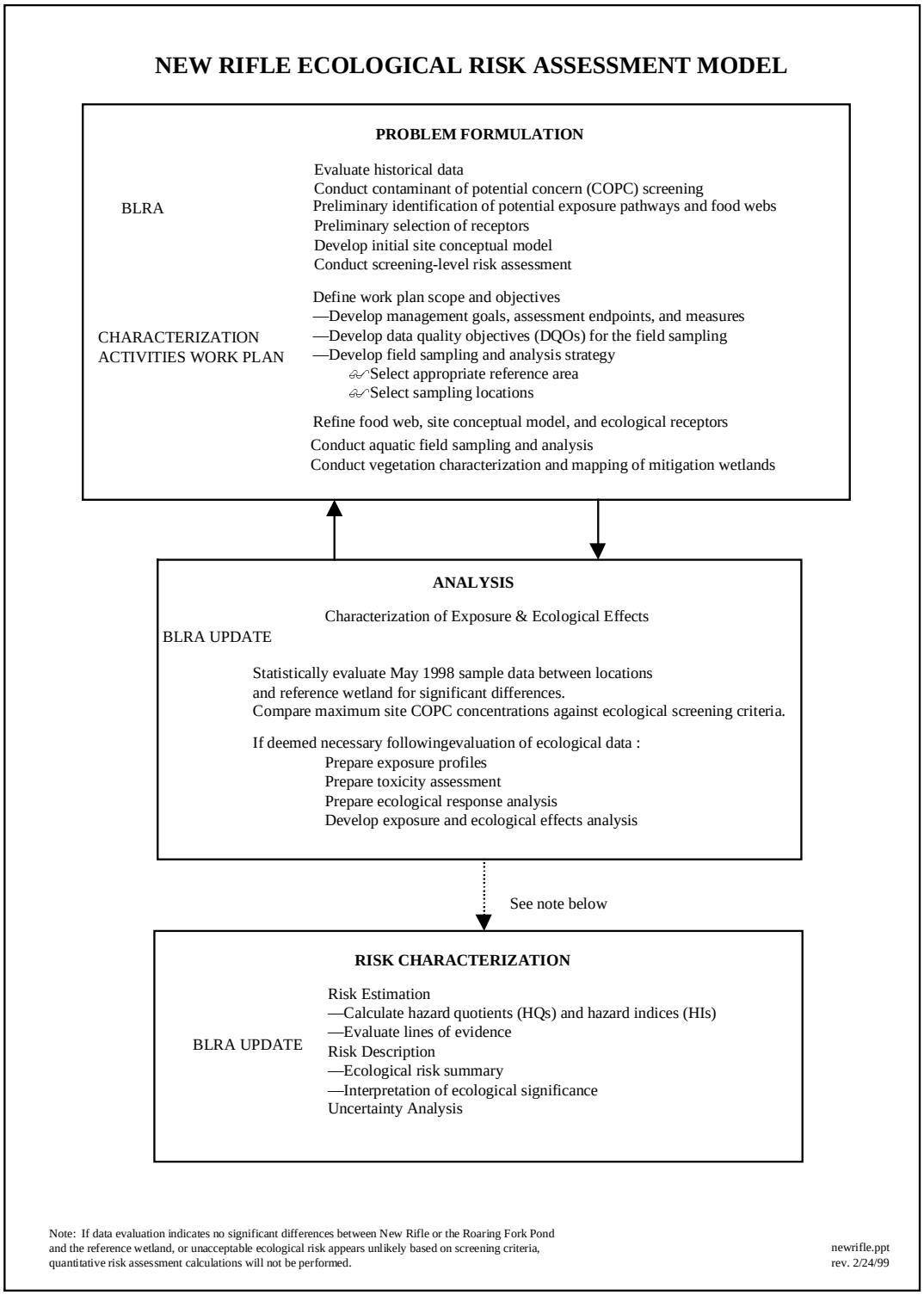

Figure 6–1. Risk Assessment Model

activities associated with the New Rifle wetlands. Section 6.2.4 briefly summarizes the methodology and results of the plant uptake study.

6.2.2.1 Ecological Risk Model

For the purpose of completing a QRA for the BLRA update, a more detailed site conceptual model was developed for the New Rifle site. ("Site" refers to both the actual New Rifle site and the relevant portions of the adjacent Roaring Fork gravel operation to the west—particularly the gravel pit pond.) To identify and evaluate all relevant exposure pathways and receptors, both the current exposure scenario and a hypothetical future exposure scenario were considered. Figure 6–2 schematically depicts the risk model and identifies contaminant sources, release and transport mechanisms, general categories of ecological receptors, and routes of exposure. Minor pathways that pose no significant risks were identified and eliminated from further consideration.

The major exposure pathways retained for further evaluation at the New Rifle Site are:

- Direct contact with contaminated abiotic media (e.g., plants in contact with contaminated sediment or surface water).
- Ingestion of abiotic media (e.g., ingestion of sediments or surface water by birds or animals).
- Ingestion of contaminated biological media (i.e., ingestion of plants or animals that have accumulated contamination through exposure to site-related contaminants).

The terrestrial habitat at the site is of limited concern because surface remediation has been completed. The site is currently not used for livestock grazing. Wildlife has access to the site through the river corridor or adjacent properties. These animals could be exposed to contamination through food items and drinking water obtained from areas with contaminated surface water and sediments. However, their occasional exposure is not expected to produce significant adverse effects. Most of the terrestrial plants at the site are shallow-rooted and unlikely to come in contact with contaminated ground water. Based on projected land and water use, the terrestrial habitat is unlikely to pose any greater concern in the future.

The aquatic habitat associated with the New Rifle wetland and the Roaring Fork pond represent the areas of greatest potential exposure. Although these areas do not currently represent significant aquatic habitats, the reconstruction of the wetland and eventual cessation of the gravel mining operation are likely to increase their viability in the future.

Both the New Rifle wetland and holding pond are fed by alluvial ground water contaminated from the former milling operation. Contaminants in the surface water are likely to interact with sediments, which may serve as contaminant sinks. Phreatophytes (plants with root systems that extend below the water table) can root in sediment and may uptake contaminants through their root systems. Receptors living in the aquatic habitat, such as aquatic invertebrates and fish, take in contaminated water and sediment when feeding and also are exposed to these media through direct contact. Birds and mammals that live in or frequent the aquatic habitat (e.g., beavers, mule deer, ducks) are most likely to contact contamination through dietary intakes of surface water, biota, and sediment.

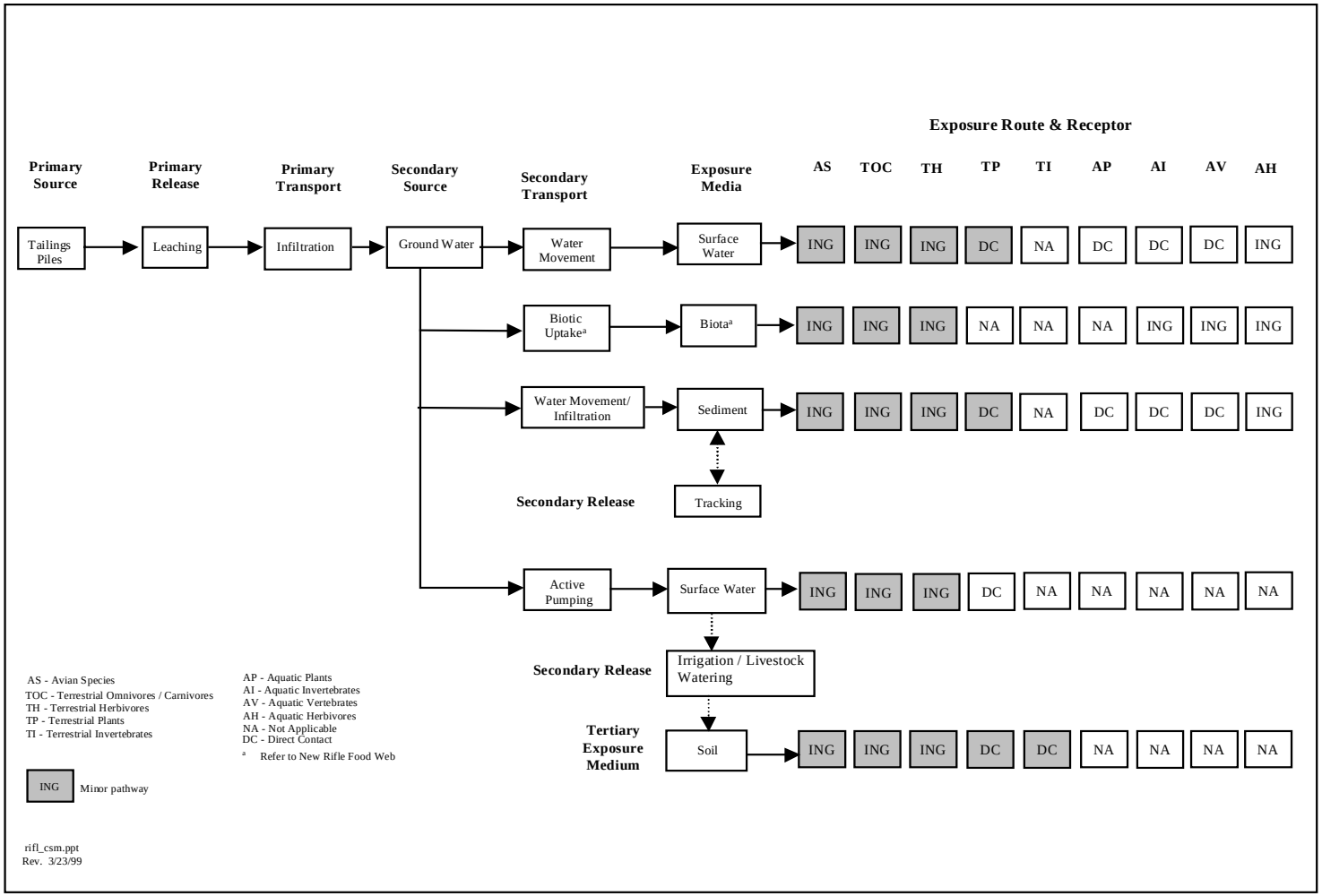

Figure 6–2. New Rifle Site Conceptual Ecological Risk Model

6.2.2.2 Methodology

The quantitative assessment of ecological risk requires identification of relevant exposure pathways, representative ecological receptors for those pathways, and contaminants and concentrations likely to be present through each pathway. To be conservative, the following potential exposure pathways were considered and evaluated quantitatively in the BLRA update:

- Surface water—ingestion and direct contact, as appropriate, by receptor
- Sediment—ingestion and direct contact, as appropriate, by receptor
- Dietary—ingestion of forage and/or prey, as appropriate, by receptor

Numerous receptor species were identified at the site through ecological surveys. However, to streamline the risk assessment process, a limited number of species were selected to evaluate site risks. The selected organisms were determined to be representative of the receptor categories for each of the ecological environments at the site that were likely to come into contact with site-related contamination and play a key role in the site food chain. A generalized food web for the New Rifle site is shown in Figure 6–3. The receptors selected were the muskrat, mule deer, great blue heron, benthic invertebrates, creek chub, and aquatic plants. Risks to muskrat, mule deer, and great blue heron were determined for ingestion of surface water, sediment, and dietary intakes. Evaluations of risk to benthic invertebrates and creek chub were based on direct exposure to surface water and sediment. Effects of contaminant uptake from sediments were predicted for aquatic plants. A separate study was undertaken to evaluate the site-specific impacts of ground water discharge to the New Rifle wetland and subsequent contaminant uptake by plants from water. The full report on this study is contained in a “Report to MACTEC–ERS: Phytotoxic Evaluation of Ammoniacal-N Contaminated Groundwater” (BRI 1999); results are described in Section 6.2.4.

The list of COPCs in Table 6–4 was further reduced during refinement of the site conceptual model for additional data collection and evaluation. Calcium, potassium, and sodium were eliminated because they are macronutrients that are not expected to have toxic effects. Silica was eliminated because of its low toxicity and its ubiquitous occurrence in the environment. Chloride exceeds chronic aquatic criteria but was inadvertently omitted from the COPC list. It will be included as an analyte for any future monitoring that may be required to address ecological risk concerns. The remaining 15 COPCs were retained as analytes for samples collected during the ERA field investigation.

Surface water and sediment samples were collected from the New Rifle wetland and the Roaring Fork pond for comparison with samples from the reference area. Reference area samples were collected from One Mile Pond located upgradient from and outside the influence of both the New and Old Rifle sites. All sample locations are shown in Figure 4–6. Data for both filtered and unfiltered surface water were evaluated. Samples of cattail stems and roots were also collected from the New Rifle wetland and the reference area to evaluate the plant component of the dietary ingestion pathway. Because of a lack of vegetation at Roaring Fork pond, concentrations of plant tissue were estimated on the basis of uptake factors calculated from results of New Rifle wetland and reference area samples and contaminant concentrations detected in Roaring Fork pond sediments. Contaminant concentrations in other biota were estimated for all locations on the

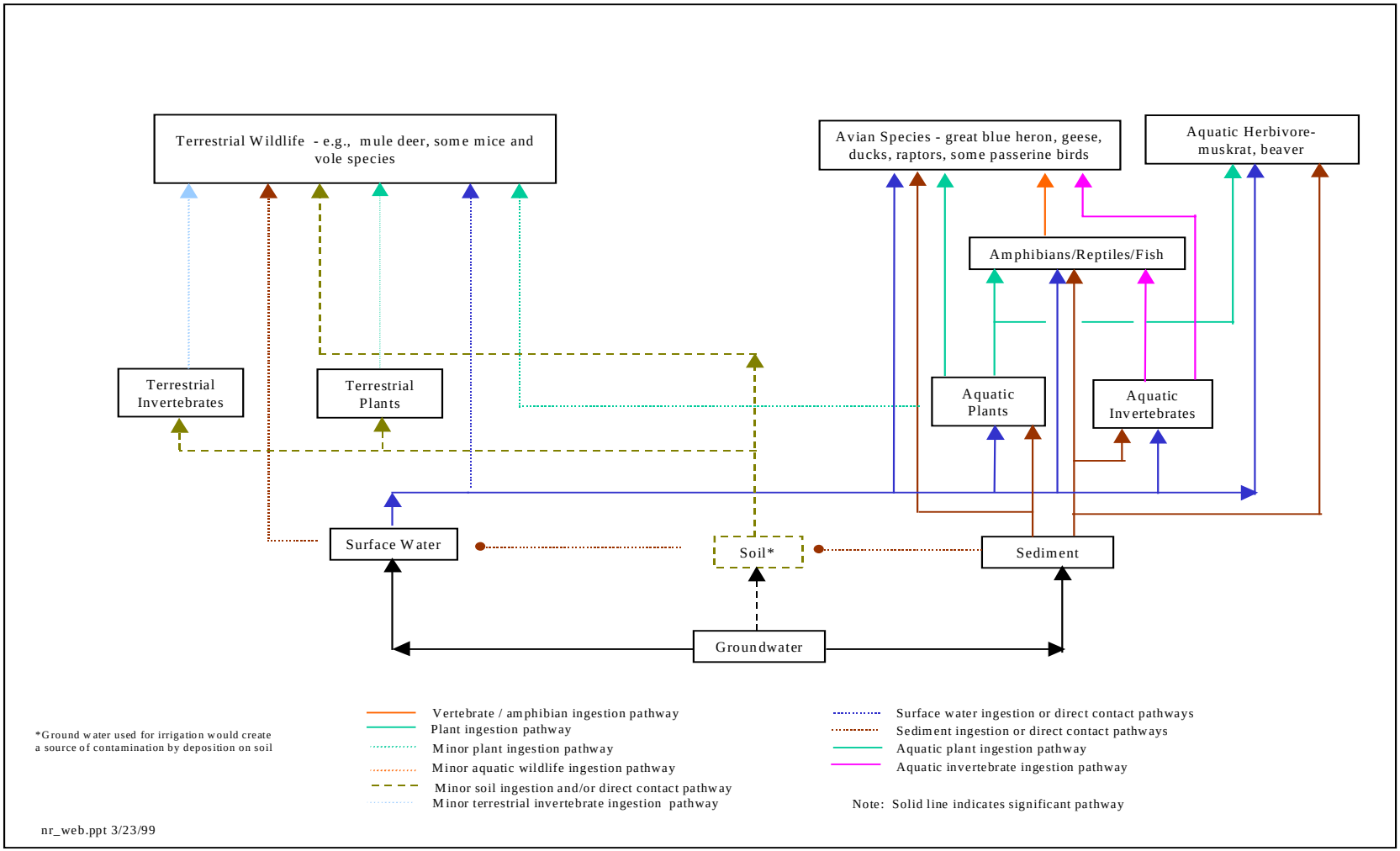

Figure 6–3. Generalized New Rifle Food Web

basis of surface water and sediment concentrations and appropriate uptake factors obtained from the literature, where available.

Summary statistics of measured COPC concentrations in relevant media are provided in Table E–1 of Appendix E (Note that ammonia was analyzed as NH_4; for subsequent risk evaluation the values were converted to N). A statistical analysis was performed comparing means of site contaminant concentrations with those of the reference area location for all relevant media. New Rifle wetland and Roaring Fork pond data sets were evaluated separately. Results of that analysis, in combination with the summary statistics, were used to screen out COPCs for each medium and location. COPCs in a given medium were eliminated from further consideration at either the New Rifle wetland or Roaring Fork pond locations if they met any of the following criteria:

- Not detected in any samples at that site for a given medium
- The maximum concentration at the reference area exceeded the maximum concentration at the site
- Means were the same based on statistical testing, but actual calculated reference area mean exceeded mean for site samples.

Note that contaminant concentrations in Table E–1, Appendix E for vegetation are reported on a dry weight basis. These concentrations are used throughout the ecological risk calculations for dietary intakes. Use of dry weight concentrations serves to overestimate risks by nearly an order of magnitude as intakes of vegetation by ecological receptors are based largely on wet weights. Vegetation at the New Rifle site consists, on the average, of nearly 90 percent water.

The statistical comparison of data resulted in a different set of COPCs for the New Rifle wetland and the Roaring Fork pond. Estimated biota concentrations also differed for these two locations. To account for these differences in subsequent quantitative risk calculations, separate sets of reference area exposure point concentrations (EPCs) were created for these two locations. EPCs used represent the UCL_{95} of the sample mean or the maximum detected concentration, whichever was lowest. Table E–2 of Appendix E contains results of the statistical testing. The EPCs used for the New Rifle wetland, Roaring Fork pond, and their respective reference areas are provided in Table E–3 of Appendix E. Reference area concentrations for both data sets are the same for abiotic media; they differ only in which COPCs are retained for further analysis. COPCs retained after the statistical screening process were ammonia, arsenic, cadmium, fluoride, iron, manganese, molybdenum, nitrate, phosphate, selenium, strontium, sulfate, uranium, vanadium, and zinc (chloride was inadvertently omitted as a COPC; it will be included as a COPC in any monitoring or characterization required in the future). Quantitative risks associated with each of these contaminants were calculated for each relevant medium. Table 6–5 summarizes the COPCs retained for QRA.

Table 6–5. COPCs Retained for Quantitative Risk Assessment

Analyte	Surface Water	Sediment	Cattails	Other Biota
Ammonia	NRW, RFP	RFP	RFP	NRW, RFP
Arsenic	NRW, RFP	NRW, RFP	RFP	NRW, RFP
Cadmium		NRW, RFP	NRW, RFP	
Fluoride	NRW, RFP			NRW, RFP
Iron	NRW	RFP	RFP	NRW
Manganese		NRW, RFP	RFP	
Molybdenum	NRW, RFP	NRW, RFP	NRW, RFP	NRW, RFP
Nitrate	NRW, RFP	NRW, RFP	RFP	NRW, RFP
Phosphate		NRW, RFP	RFP	
Selenium	RFP	RFP	NRW, RFP	RFP
Strontium	NRW, RFP	NRW	NRW	NRW, RFP
Sulfate	RFP	RFP	RFP	RFP
Uranium	NRW, RFP	RFP	RFP	NRW, RFP
Vanadium	NRW, RFP	NRW, RFP	RFP	NRW, RFP
Zinc		NRW, RFP	RFP	

NRW = New Rifle wetland
RFP = Roaring Fork pond

Table 6–6 summarizes the QRA process, which followed methodology similar to that established for conducting human health risk assessments (EPA 1989b) and is described briefly below. Data used to calculate quantitative risks are included in Tables E–2 through E–8 of Appendix E; risk equations used in the calculations are also provided in the Appendix along with a brief description of those calculations. Further discussion is provided in Calculation Set U0060400.

Contaminant intakes for muskrat, mule deer, and great blue heron were estimated using exposure parameters from the literature and site-specific EPCs. Intake calculations were performed for both a reasonable worst-case scenario (reasonable maximum exposure, or RME) and an average exposure scenario (central tendency, or CT). As a conservative measure, it was assumed that all food and water ingested/contacted by the receptors was from on site and was contaminated.

To evaluate potential risks associated with contaminant intakes by muskrat, mule deer, and great blue heron, intakes were compared against toxicity reference values (TRVs) obtained from a variety of references from the literature. In most cases TRVs were available only for nonsite species and had to be adjusted using uncertainty factors for application to site-specific receptors (see Calculation Set U0060400). (Application of uncertainty factors always results in lower, more conservative TRVs). The calculated TRVs represent intakes of contaminants that would result in no adverse effects to the receptor (NOAEL, or no observed adverse effect level) as well as intakes representing the lowest concentration at which adverse effects can be expected (LOAEL, or lowest observed adverse effect level). Generally, the combination of exposure parameters and TRVs can be interpreted as follows (DOE 1998b):

Measure of Exposure	Measure of Toxicity	Risk Estimate
CT	NOAEL	Average exposed individuals
RME	NOAEL	Highly exposed individuals
CT	LOAEL	Average exposed populations
RME	LOAEL	Highly exposed populations

Table 6–6. Methodology For New Rifle Quantitative Ecological Risk Assessment

Step 1. Determine COPCs for relevant exposure pathways and receptors (Table E–2, Appendix E)

--initial list used current and historical background concentrations; retained COPCs significantly elevated above background

--using 1998 data only, performed more detailed statistical testing to determine if differences existed between reference area and site-related sample means; retained COPCs if :

- site means > reference means,
- if statistical testing showed no significant differences between means, but actual site mean > reference area mean,
- if data not adequate for statistical testing, but site data obviously greater than reference area (e.g., all detects for site data and non-detects for reference area data)

Step 2. Determine Exposure Point Concentrations (EPCs) for each COPC (Table E–3, Appendix E)

--used maximum value or UCL_{95} (whichever lower) for sediment, surface water

--muskrat, mule deer dietary EPCs

- New Rifle wetland – used measured concentrations in cattail stems, leaves for COPCs remaining after statistical testing
- Roaring Fork pond – used results of New Rifle wetland & reference area to calculate BAFs; used sediment concentrations at Roaring Fork pond to predict vegetation concentrations for COPCs in sediments (BAFs in Table E–4, Appendix E; EPCs in Table E–3)

--great blue heron dietary EPCs

- Used surface water COPC concentrations & literature BCFs to predict tissue concentrations in frogs, fish, etc. (BCFs in Table E–4, Appendix E; EPCs in Table E–3)

Step 3. Determine intakes for muskrat, mule deer, and great blue heron

--used literature RME and CT exposure parameters and EPCs for each relevant medium (Table E–5, Appendix E)

--calculated each intake from sediment, surface water, and dietary intake in mg/kg bw/day (Table E–6, Appendix E)

Step 4. Toxicity evaluation – Determine appropriate toxicity reference values (TRVs) for each receptor

--used literature values for muskrat, mule deer, great blue heron for NOAEL and LOAEL; extrapolated from other species by application of uncertainty factors (TRVs in mg/kg bw/day) (Table E–7, Appendix E)

--for benthic invertebrates, creek chub, and aquatic plants, used AWQC and SQC as reference values (for plants evaluated sediment only) instead of using intake calculations (Table E–8, Appendix E)

Step 5. Risk Characterization – Determine HQs and HIs (Methodology, Appendix E)

--for muskrat, mule deer, and great blue heron: Intake/TRV = HQ; ΣHQ for given contaminant = HI; determined for RME, CT scenarios And NOAEL and LOAEL TRVs (Results, Table E–9, Appendix E)

--for benthic invertebrates, creek chub, aquatic plants: EPC_{sed}/SQC, EPC_{sw}/AWQC = HQ; ΣHQ for given contaminant = HI (Table E–9, Appendix E)

Although the development of a site compliance strategy focuses mainly on providing protection at the population level, the discussion of results presented here centers mainly on the most conservative scenario using RME and NOAEL data.

For benthic invertebrates and fish (represented by the creek chub), contaminant intakes were not calculated, but exposure was evaluated through the use of aquatic water quality criteria (AWQC) and sediment quality criteria (SQC) from the literature, where available. These criteria are not species-specific but are deemed to be protective to the majority of aquatic life. Both chronic and acute benchmarks were evaluated and assumed to be analogous to NOAEL and LOAEL concentrations, respectively. Because exposure intakes were not calculated, the same data were used to evaluate RME and CT scenarios. Uptakes were also not calculated for aquatic plants, and TRVs represent sediment concentrations corresponding to NOAEL and LOAEL concentrations. Again, because uptakes were not determined, the same data were used for RME and CT scenarios.

To determine the potential magnitude of risks associated with a particular contaminant and exposure pathway, HQs were calculated. For muskrat, mule deer, and great blue heron, HQs represent the ratio of the calculated intake of a contaminant for a given receptor and exposure pathway to the TRV for that receptor, contaminant, and pathway. HQs were calculated using both NOAEL and LOAEL TRVs. For benthic invertebrates and fish, HQs are the ratio of the EPCs for each contaminant in surface water and sediment to the appropriate AWQC or SQC, respectively (both chronic and acute, if available). For aquatic plants, HQs are the ratio of each sediment EPC to its respective TRV. For each receptor, summing the HQs for a given contaminant for all pathways provides a HI for that contaminant. HIs or HQs greater than 1 may represent an unacceptable risk.

6.2.2.3 Results of QRA

Tables 6–7 through 6–10 present the results of the quantitative ERA as represented by calculated HQs and HIs for all contaminants, pathways, and receptors evaluated. Results are provided for RME exposure scenarios using both NOAEL and LOAEL benchmarks. (For CT exposure scenario results, see Table E–9 in Appendix E.) Note that available data were incomplete and that it was not possible to accurately assess potential risks posed to some receptors for a number of contaminant and pathway combinations. The shaded values in Tables 6–7 through 6–10 represent HIs that could not be determined or that are based on only partial exposure data. The worst-case HQs and HIs are those for the RME scenario evaluated using NOAEL benchmarks.

Table E–10 in Appendix E highlights the risk drivers, or HQs and HIs exceeding 1, for all pathways and receptors evaluated.

To evaluate risks associated with the New Rifle site locations against those posed by ambient background conditions, ratios of site risks to background risks were determined to assess relative risks (RR). An RR is the ratio of an HQ or HI for a given site contaminant, receptor, and exposure pathway to the HQ or HI for that same contaminant, receptor, and exposure pathway for the respective reference area. If an RR is less than or equal to 1, site-related contamination is of no greater risk to an ecological receptor than is natural background. Tables 6–11 and 6–12 present the RRs for all contaminants, pathways, and receptors evaluated that had an HQ or HI greater than 1 ***and*** that exceeded the respective HQ or HI calculated for the appropriate reference area (i.e., RR $\geq$ 1 using NOAEL-RME scenario). Shaded values indicate relative risks where the

Table 6–7. New Rifle Wetland Ecological Risks (NOAEL-RME)

	NOAEL-Based HQ's for Sediment Ingestion - RME			NOAEL-Based HQ's for Dietary Ingestion - RME			NOAEL-Based HQ's for SW Ingestion - RME			NOAEL-Based HI - RME		
Parameter	**Muskrat**	**Mule Deer**	**Great Blue Heron**	**Muskrat**	**Mule Deer**	**Great Blue Heron**	**Muskrat**	**Mule Deer**	**Great Blue Heron**	**Muskrat**	**Mule Deer**	**Great Blue Heron**
Ammonia (as N)	NA	NA	NA	NA	NA	no TRV	1.23E-05	1.76E-07	no TRV	1.23E-05	1.76E-07	no TRV
Arsenic	4.15E-02	9.61E-03	1.66E-02	NA	NA	1.03E-02	6.35E-03	2.54E-04	1.38E-04	4.79E-02	9.86E-03	2.70E-02
Cadmium	3.02E-02	5.22E-03	1.25E-01	1.31E+00	1.81E-01	NA	NA	NA	NA	1.34E+00	1.86E-01	1.25E-01
Fluoride	NA	NA	NA	NA	NA	no BCF	8.13E-01	3.24E-02	4.14E-02	8.13E-01	3.24E-02	4.14E-02
Iron	NA	NA	NA	NA	NA	no BCF	2.91E-01	6.96E-03	7.02E-04	2.91E-01	6.96E-03	7.02E-04
Manganese	3.55E-02	1.54E-02	1.91E-02	NA	NA	NA	NA	NA	NA	3.55E-02	1.54E-02	1.91E-02
Molybdenum	1.03E-01	1.10E-02	2.35E-03	18.6	1.89	no BCF	7.07E-01	1.30E-02	8.73E-04	19.41	1.914	3.22E-03
Nitrate	7.45E-02	1.04E-02	2.53E-01	NA	NA	no BCF	1.15E-02	2.75E-04	2.10E-03	8.60E-02	1.07E-02	2.55E-01
Phosphate	5.69E-04	7.88E-05	5.30E-04	NA	NA	NA	NA	NA	NA	5.69E-04	7.88E-05	5.30E-04
Selenium	NA	NA	NA	6.65E-01	1.15E-01	NA	NA	NA	NA	6.65E-01	1.15E-01	NA
Strontium	4.70E-02	6.52E-03	No TRV	2.35E+00	2.43E-01	no TRV	4.18E-02	1.00E-03	no TRV	2.44E+00	2.51E-01	no TRV
Sulfate	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Uranium	NA	NA	NA	NA	NA	no BCF	1.09E-02	4.34E-04	7.33E-04	1.09E-02	4.34E-04	7.33E-04
Vanadium	1.21E+00	2.93E-01	2.31E+01	NA	NA	no BCF	1.96E-02	8.20E-04	2.04E-02	1.23E+00	2.94E-01	2.31E+01
Zinc	4.64E-03	1.95E-01	4.29E-02	NA	NA	NA	NA	NA	NA	4.64E-03	1.95E-01	4.29E-02

NA = not a COPC
no BCF = no bioconcentration factor to calculate concentrations for food in diet
no TRV = no toxicity reference data
Shaded values represent HIs based on partial intakes; total intake could not be determined due to lack of data

	NOAEL-Based HQ's for Sediment Direct Contact			NOAEL-Based HQ's for Direct Contact - Surface		NOAEL-Based HIs Summed Across Pathways		
Parameter	**Benthic Invertebrates**	**Creek Chub**	**Aquatic Plants**	**Benthic Invertebrates**	**Creek Chub**	**Benthic Invertebrates**	**Creek Chub**	**Aquatic Plants**
Ammonia (as N)	NA	NA	NA	1.24E-02	1.24E-02	1.24E-02	1.24E-02	NA
Arsenic	1.72E+00	1.72E+00	4.13E-01	4.53E-02	4.53E-02	1.77E+00	1.77E+00	4.13E-01
Cadmium	1.23	1.23	1.48E-02	NA	NA	1.23	1.23	1.48E-02
Fluoride	NA	NA	NA	2.40E+02	2.40E+02	2.40E+02	2.40E+02	NA
Iron	NA	NA	NA	2.16E-02	2.16E-02	2.16E-02	2.16E-02	NA
Manganese	7.56E-01	7.56E-01	6.95E-01	NA	NA	7.56E-01	7.56E-01	6.95E-01
Molybdenum	no TRV	no TRV	no TRV	3.23E-01	3.23E-01	3.23E-01	3.23E-01	no TRV
Nitrate	no TRV	no TRV	no TRV	no TRV	no TRV	no TRV	no TRV	no TRV
Phosphate	no TRV	no TRV	no TRV	NA	NA	no TRV	no TRV	no TRV
Selenium	NA	NA	NA	NA	NA	NA	NA	NA
Strontium	no TRV	no TRV	no TRV	8.90E-01	8.90E-01	8.90E-01	8.90E-01	no TRV
Sulfate	NA	NA	NA	NA	NA	NA	NA	NA
Uranium	NA	NA	NA	1.51E+01	1.51E+01	1.51E+01	1.51E+01	NA
Vanadium	no TRV	no TRV	1.66E+01	4.63E-01	4.63E-01	4.63E-01	4.63E-01	1.66E+01
Zinc	4.29E-01	4.29E-01	1.36E-01	NA	NA	4.29E-01	4.29E-01	1.36E-01

NA = not a COPC
no TRV = no toxicity reference value
Shaded values represent partial HIs or HIs that could not be determined based on lack of toxicity data

Table 6–8. New Rifle Wetland - Ecological Risks (LOAEL-RME)

	LOAEL-Based HQ's for Sediment Ingestion - RME			LOAEL-Based HQ's for Dietary Ingestion - RME			LOAEL-Based HQ's for SW Ingestion - RME			LOAEL-Based HI - RME		
Parameter	Muskrat	Mule Deer	Great Blue Heron	Muskrat	Mule Deer	Great Blue Heron	Muskrat	Mule Deer	Great Blue Heron	Muskrat	Mule Deer	Great Blue Heron
Ammonia (as N)	NA	NA	no TRV	NA	NA	no TRV	6.15E-06	8.82E-08	no TRV	6.15E-06	8.82E-08	no TRV
Arsenic	7.03E-03	1.62E-03	5.50E-03	NA	NA	3.44E-03	1.08E-03	4.28E-05	4.60E-05	8.11E-03	1.66E-03	8.99E-03
Cadmium	7.76E-04	1.08E-04	8.32E-03	3.37E-02	3.73E-03	NA	NA	NA	NA	3.45E-02	3.84E-03	8.32E-03
Fluoride	NA	NA	no TRV	NA	NA	no BCF	1.60E-01	6.35E-03	no TRV	1.60E-01	6.35E-03	no TRV
Iron	NA	NA	NA	NA	NA	no BCF	1.37E-02	3.27E-04	3.51E-04	1.37E-02	3.27E-04	3.51E-04
Manganese	1.15E-02	no TRV	9.54E-03	NA	NA	NA	NA	NA	NA	1.15E-02	no TRV	9.54E-03
Molybdenum	2.06E-02	2.29E-03	1.57E-03	3.73E+00	3.93E-01	no BCF	1.41E-01	2.70E-03	5.83E-04	3.89E+00	3.98E-01	2.15E-03
Nitrate	3.43E-03	3.80E-04	7.20E-03	NA	NA	no BCF	5.23E-04	1.00E-05	5.98E-05	3.95E-03	3.90E-04	7.26E-03
Phosphate	no TRV	no TRV	no TRV	no TRV	no TRV	NA	NA	NA	NA	no TRV	no TRV	no TRV
Selenium	NA	NA	NA	4.99E-01	8.61E-02	NA	NA	NA	NA	4.99E-01	8.61E-02	NA
Strontium	4.70E-02	6.52E-03	No TRV	2.35E+00	2.43E-01	no TRV	4.18E-02	1.00E-03	no TRV	2.44E+00	2.51E-01	no TRV
Sulfate	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA	NA
Uranium	NA	NA	NA	NA	NA	no BCF	4.36E-03	1.74E-04	1.47E-05	4.36E-03	1.74E-04	1.47E-05
Vanadium	2.42E-01	5.58E-02	1.54E+00	NA	NA	no BCF	3.92E-03	1.56E-04	1.36E-03	2.46E-01	5.60E-02	1.54E+00
Zinc	2.32E-03	1.01E-01	6.14E-03	NA	NA	NA	NA	NA	NA	2.32E-03	1.01E-01	6.14E-03

NA = not a COPC
no BCF = no bioconcentration factor to calculate concentrations for food in diet
ND = not determined due to lack of data
Shaded values represent HIs based on partial intakes; total intake could not be determined due to lack of data

	LOAEL-Based HQ's for Sediment Direct Contact			LOAEL-Based HQ's for Direct Contact - Surface		LOAEL-Based HIs Summed Across Pathways		
Parameter	Benthic Invertebrates	Creek Chub	Aquatic Plants	Benthic Invertebrates	Creek Chub	Benthic Invertebrates	Creek Chub	Aquatic Plants
Ammonia (as N)	NA	NA	NA	1.95E-03	1.95E-03	1.95E-03	1.95E-03	NA
Arsenic	1.72E+00	1.72E+00	no TRV	2.39E-02	2.39E-02	1.74E+00	1.74E+00	no TRV
Cadmium	1.23E+00	1.23E+00	7.39E-03	NA	NA	1.23E+00	1.23E+00	7.39E-03
Fluoride	NA	NA	NA	1.51E+01	1.51E+01	1.51E+01	1.51E+01	NA
Iron	NA	NA	NA	no TRV	no TRV	no TRV	no TRV	NA
Manganese	7.56E-01	7.56E-01	no TRV	NA	NA	7.56E-01	7.56E-01	no TRV
Molybdenum	no TRV	no TRV	no TRV	no TRV	no TRV	no TRV	no TRV	no TRV
Nitrate	no TRV	no TRV	no TRV	no TRV	no TRV	no TRV	no TRV	no TRV
Phosphate	no TRV	no TRV	no TRV	NA	NA	no TRV	no TRV	no TRV
Selenium	NA	NA	NA	NA	NA	NA	NA	NA
Strontium	no TRV	no TRV	no TRV	8.90E-02	8.90E-02	8.90E-02	8.90E-02	no TRV
Sulfate	NA	NA	NA	NA	NA	NA	NA	NA
Uranium	NA	NA	NA	1.46E+00	1.46E+00	1.46E+00	1.46E+00	NA
Vanadium	no TRV	no TRV	no TRV	no TRV	no TRV	no TRV	no TRV	no TRV
Zinc	4.29E-01	4.29E-01	no TRV	NA	NA	4.29E-01	4.29E-01	no TRV

NA = not a COPC
no TRV = no toxicity reference value
Shaded values represent partial HIs or HIs that could not be determined based on lack of toxicity data

Table 6–9. Roaring Fork Pond - Ecological Risks (NOAEL-RME)

	NOAEL-Based HQ's for Sediment Ingestion - RME			NOAEL-Based HQ's for Dietary Ingestion - RME			NOAEL-Based HQ's for SW Ingestion - RME			NOAEL-Based HI - RME		
Parameter	**Muskrat**	**Mule Deer**	**Great Blue Heron**	**Muskrat**	**Mule Deer**	**Great Blue Heron**	**Muskrat**	**Mule Deer**	**Great Blue Heron**	**Muskrat**	**Mule Deer**	**Great Blue Heron**
Ammonia (as N)	2.08E-03	1.73E-04	no TRV	No BAF	No BAF	no TRV	1.56E-02	2.24E-04	no TRV	1.77E-02	3.97E-04	no TRV
Arsenic	2.87E-02	6.65E-03	1.15E-02	3.28E+00	5.67E-01	5.33E-03	3.28E-03	1.31E-04	7.13E-05	3.31E+00	5.74E-01	1.69E-02
Cadmium	3.02E-02	5.24E-03	1.25E-01	1.01E+01	1.14E+00	NA	NA	NA	NA	1.01E+01	1.14E+00	NA
Fluoride	NA	NA	NA	NA	NA	no BCF	5.89E-01	2.35E-02	3.00E-02	5.89E-01	2.35E-02	3.00E-02
Iron	2.37E+01	3.29E+00	1.05E+00	1.36E+03	3.55E+01	NA	NA	NA	NA	1.38E+03	3.88E+01	1.05E+00
Manganese	4.29E-02	1.86E-02	2.31E-02	5.58E+00	2.23E+00	NA	NA	NA	NA	5.62E+00	2.25E+00	NA
Molybdenum	1.79E-01	1.90E-02	4.06E-03	1.08E+01	7.93E-01	no BCF	5.05E+00	9.29E-02	6.25E-03	1.60E+01	9.05E-01	1.03E-02
Nitrate	1.74E-01	2.42E-02	5.84E-01	No BAF	No BAF	no BCF	7.34E+01	1.75E+00	13.4	73.574	1.77E+00	1.40E+01
Phosphate	4.54E-04	6.30E-05	4.24E-04	No BAF	No BAF	NA	NA	NA	NA	4.54E-04	6.30E-05	4.24E-04
Selenium	4.20E-03	9.77E-04	1.18E-02	No BAF	No BAF	1.04E-01	2.57E-03	1.03E-04	3.92E-04	6.77E-03	1.08E-03	1.16E-01
Strontium	NA	NA	NA	NA	NA	no TRV	1.03E-01	2.47E-03	no TRV	1.03E-01	2.47E-03	no TRV
Sulfate	7.35E+00	1.02E+00	6.16E-02	No BAF	No BAF	no BCF	1.45E+03	3.48E+01	6.63E-01	1.46E+03	3.58E+01	7.25E-01
Uranium	9.07E-03	2.10E-03	1.12E-02	3.43E-01	1.23E-02	no BCF	4.67E-02	1.86E-03	3.14E-03	3.99E-01	1.63E-02	1.43E-02
Vanadium	1.18E+00	2.86E-01	2.25E+01	7.16E+01	2.94E+00	no BCF	1.96E-02	8.20E-04	2.04E-02	7.28E+01	3.23E+00	2.25E+01
Zinc	7.99E-03	3.35E-01	7.40E-02	6.63E-01	1.71E+01	NA	NA	NA	NA	6.71E-01	1.74E+01	7.40E-02

NA = not a COPC
no BCF = no bioconcentration factor to calculate concentrations for food in diet
ND = not determined due to lack of data
Shaded values represent HIs based on partial intakes; total intake could not be determined due to lack of data

	NOAEL-Based HQ's for Sediment Direct Contact			NOAEL-Based HQ's for Direct Contact - Surface		NOAEL-Based HIs Summed Across Pathways		
Parameter	**Benthic Invertebrates**	**Creek Chub**	**Aquatic Plants**	**Benthic Invertebrates**	**Creek Chub**	**Benthic Invertebrates**	**Creek Chub**	**Aquatic Plants**
Ammonia (as N)	no TRV	no TRV	no TRV	2.92E+00	2.92E+00	2.92E+00	2.92E+00	no TRV
Arsenic	1.19E+00	1.19E+00	2.86E-01	2.95E-02	2.95E-02	1.22E+00	1.22E+00	2.86E-01
Cadmium	1.23E+00	1.23E+00	1.48E-02	NA	NA	1.23E+00	1.23E+00	1.48E-02
Fluoride	NA	NA	NA	1.75E+02	1.75E+02	1.75E+02	1.75E+02	NA
Iron	9.12E-01	9.12E-01	no TRV	NA	NA	9.12E-01	NA	no TRV
Manganese	9.14E-01	9.14E-01	8.41E-01	NA	NA	9.14E-01	9.14E-01	8.41E-01
Molybdenum	no TRV	no TRV	no TRV	2.11E+00	2.11E+00	2.11E+00	2.11E+00	no TRV
Nitrate	no TRV	no TRV	no TRV	no TRV	no TRV	no TRV	no TRV	no TRV
Phosphate	no TRV	no TRV	no TRV	NA	NA	no TRV	no TRV	no TRV
Selenium	no TRV	no TRV	3.46E-01	6.54E-01	6.54E-01	6.54E-01	6.54E-01	3.46E-01
Strontium	NA	NA	NA	2.30E+00	2.30E+00	2.30E+00	2.30E+00	NA
Sulfate	2.27E+01	2.27E+01	no TRV	9.93E+03	9.93E+03	9.95E+03	9.95E+03	no TRV
Uranium	no TRV	no TRV	no TRV	6.08E+01	6.08E+01	6.08E+01	6.08E+01	no TRV
Vanadium	no TRV	no TRV	1.62E+01	4.84E-01	4.84E-01	4.84E-01	4.84E-01	1.62E+01
Zinc	7.40E-01	7.40E-01	2.34E-01	NA	NA	7.40E-01	7.40E-01	2.34E-01

NA = not a COPC
no TRV = no toxicity reference value
Shaded values represent partial HIs or HIs that could not be determined based on lack of toxicity data

Table 6–10. Roaring Fork Pond - Ecological Risks (LOAEL-RME)

	LOAEL-Based HQ's for Sediment Ingestion - RME			LOAEL-Based HQ's for Dietary Ingestion - RME			LOAEL-Based HQ's for SW Ingestion - RME			LOAEL-Based HI - RME		
Parameter	Muskrat	Mule Deer	Great Blue Heron	Muskrat	Mule Deer	Great Blue Heron	Muskrat	Mule Deer	Great Blue Heron	Muskrat	Mule Deer	Great Blue Heron
Ammonia (as N)	1.04E-03	8.65E-05	no TRV	No BAF	No BAF	no TRV	7.80E-03	1.12E-04	no TRV	8.84E-03	1.99E-04	no TRV
Arsenic	4.86E-03	1.12E-03	3.83E-03	5.56E-01	9.58E-02	1.78E-03	5.55E-04	2.21E-05	2.38E-05	5.61E-01	9.69E-02	5.63E-03
Cadmium	7.77E-04	1.08E-04	8.33E-03	2.59E-01	2.35E-02	NA	NA	NA	NA	2.59E-01	2.35E-02	8.33E-03
Fluoride	NA	NA	NA	NA	NA	no TRV	1.16E-01	4.61E-03	no TRV	1.16E-01	4.61E-03	no TRV
Iron	1.12E+00	1.55E-01	5.26E-01	6.42E+01	1.67E+00	NA	NA	NA	NA	6.53E+01	1.83E+00	5.26E-01
Manganese	1.39E-02	no TRV	1.15E-02	1.81E+00	no TRV	NA	NA	NA	NA	1.82E+00	no TRV	1.15E-02
Molybdenum	3.57E-02	3.96E-03	2.71E-03	2.17E+00	1.65E-01	no BCF	1.01E+00	1.93E-02	4.16E-03	3.22E+00	1.88E-01	6.87E-03
Nitrate	7.93E-03	8.79E-04	1.67E-02	No BAF	No BAF	no BCF	3.34E+00	6.38E-02	3.81E-01	3.35E+00	6.47E-02	3.98E-01
Phosphate	no TRV	no TRV	no TRV	no TRV	no TRV	NA	NA	NA	NA	no TRV	no TRV	no TRV
Selenium	3.18E-03	7.34E-04	5.89E-03	No BAF	No BAF	5.22E-02	1.94E-03	7.74E-05	1.96E-04	5.12E-03	8.11E-04	5.83E-02
Strontium	NA	NA	NA	NA	NA	no TRV	1.03E-01	2.47E-03	no TRV	1.03E-01	2.47E-03	no TRV
Sulfate	2.44E+00	3.38E-01	no TRV	No BAF	No BAF	no TRV	4.83E+02	1.15E+01	no TRV	4.85E+02	1.18E+01	no TRV
Uranium	3.63E-03	8.38E-04	2.24E-04	1.37E-01	4.93E-03	no BCF	1.87E-02	7.45E-04	6.28E-05	1.59E-01	6.51E-03	2.87E-04
Vanadium	2.36E-01	5.45E-02	1.50E+00	1.43E+01	5.60E-01	no BCF	3.92E-03	1.56E-04	1.36E-03	1.45E+01	6.15E-01	1.50E+00
Zinc	3.99E-03	1.74E-01	1.06E-02	3.32E-01	8.88E+00	NA	NA	NA	NA	3.36E-01	9.05E+00	1.06E-02

NA = not a COPC
no BCF = no bioconcentration factor to calculate concentrations for food in diet
ND = not determined due to lack of data
Shaded values represent HIs based on partial intakes; total intake could not be determined due to lack of data

	LOAEL-Based HQ's for Sediment Direct Contact			LOAEL-Based HQ's for Direct Contact - Surface		LOAEL-Based HIs Summed Across Pathways		
Parameter	Benthic Invertebrates	Creek Chub	Aquatic Plants	Benthic Invertebrates	Creek Chub	Benthic Invertebrates	Creek Chub	Aquatic Plants
Ammonia (as N)	no TRV	no TRV	no TRV	4.60E-01	4.60E-01	4.60E-01	4.60E-01	no TRV
Arsenic	1.19E+00	1.19E+00	no TRV	1.56E-02	1.56E-02	1.21E+00	1.21E+00	no TRV
Cadmium	1.23E+00	1.23E+00	7.41E-03	NA	NA	1.23E+00	1.23E+00	7.41E-03
Fluoride	NA	NA	NA	1.10E+01	1.10E+01	1.10E+01	1.10E+01	NA
Iron	9.12E-01	9.12E-01	no TRV	NA	NA	9.12E-01	NA	no TRV
Manganese	9.14E-01	9.14E-01	no TRV	NA	NA	9.14E-01	9.14E-01	no TRV
Molybdenum	no TRV	no TRV	no TRV	no TRV	no TRV	no TRV	no TRV	no TRV
Nitrate	no TRV	no TRV	no TRV	no TRV	no TRV	no TRV	no TRV	no TRV
Phosphate	no TRV	no TRV	no TRV	NA	NA	no TRV	no TRV	no TRV
Selenium	no TRV	no TRV	no TRV	1.63E-01	1.63E-01	1.63E-01	1.63E-01	no TRV
Strontium	NA	NA	NA	2.30E-01	2.30E-01	2.30E-01	2.30E-01	NA
Sulfate	2.27E+01	2.27E+01	no TRV	9.93E+02	9.93E+02	1.02E+03	NA	no TRV
Uranium	no TRV	no TRV	no TRV	5.85E+00	5.85E+00	5.85E+00	5.85E+00	no TRV
Vanadium	no TRV	no TRV	no TRV	no TRV	no TRV	no TRV	no TRV	no TRV
Zinc	7.40E-01	7.40E-01	no TRV	NA	NA	7.40E-01	7.40E-01	no TRV

NA = not a COPC
no TRV = no toxicity reference value
Shaded values represent partial HIs or HIs that could not be determined based on lack of toxicity data

Table 6–11. New Rifle Wetland—Relative Ecological Risks (NOAEL-RME)

Table 6-11 (cont.) New Rifle Wetland - Relative Ecological Risks (NOAEL-RME)

	NOAEL-Based RR's for Sediment Direct Contact			NOAEL-Based RR's for Direct Contact - Surface Water		NOAEL-Based RRs Summed Across Pathways		
Parameter	Benthic Invertebrates	Creek Chub	Aquatic Plants	Benthic Invertebrates	Creek Chub	Benthic Invertebrates	Creek Chub	Aquatic Plants
Ammonia (as N)								
Arsenic	1.5	1.5				1.5	1.5	
Cadmium	4.6	4.6				4.6	4.6	
Fluoride				1.8	1.8	1.8	1.8	
Iron								
Manganese								
Molybdenum								
Nitrate								
Phosphate								
Selenium								
Strontium								
Sulfate								
Uranium				4.9	4.9	4.9	4.9	
Vanadium			1.0					
Zinc								

Table 6–12. Roaring Fork Pond—Relative Ecological Risks (NOAEL-RME)

	NOAEL-Based RR's for Sediment Ingestion - RME			NOAEL-Based RR's for Dietary Ingestion - RME			NOAEL-Based RR's for SW Ingestion - RME			NOAEL-Based RR - RME Summed across pathways		
Parameter	**Muskrat**	**Mule Deer**	**Great Blue Heron**	**Muskrat**	**Mule Deer**	**Great Blue Heron**	**Muskrat**	**Mule Deer**	**Great Blue Heron**	**Muskrat**	**Mule Deer**	**Great Blue Heron**
Ammonia (as N)												
Arsenic				1.0						1.0		
Cadmium				4.6	4.6					4.6	4.6	
Fluoride												
Iron	1.1	1.1	1.1	1.1	1.1					1.1	1.1	1.1
Manganese				1.3	1.3					1.3	1.3	
Molybdenum				5.8			368.6			8.5		
Nitrate							5691.0	5691.0	5691.0	5344.0	4146.0	2649.0
Phosphate												
Selenium												
Strontium												
Sulfate	1.6	1.6					9.9	9.9		9.7	8.7	
Uranium												
Vanadium	1.0		1.0	1.0	1.0					1.0	1.0	1.0
Zinc					1.8						1.8	

Shaded boxes = LOAEL HQ or HI also > 1

	NOAEL-Based RR's for Sediment Direct Contact			NOAEL-Based RR's for Direct Contact - Surface		NOAEL-Based RRs Summed Across Pathways		
Parameter	**Benthic Invertebrates**	**Creek Chub**	**Aquatic Plants**	**Benthic Invertebrates**	**Creek Chub**	**Benthic Invertebrates**	**Creek Chub**	**Aquatic Plants**
Ammonia (as N)				12319	12319	12319	12319	
Arsenic	1.0	1.0				1.0	1.0	
Cadmium	4.6	4.6				4.6	4.6	
Fluoride				1.3	1.3	1.3	1.3	
Iron								
Manganese								
Molybdenum				214	214	214	214	
Nitrate								
Phosphate								
Selenium								
Strontium				2.9	2.9	2.9	2.9	
Sulfate	1.6	1.6		10.0	10.0	9.9	9.9	
Uranium				19.5	19.5	19.5	19.5	
Vanadium			1.0					1.0
Zinc								

Shaded boxes = LOAEL HQ or HI also > 1

HQ or HI exceed 1 for the LOAEL-RME scenario as well. [Note that RRs are the same for both NOAEL-RME and LOAEL-RME scenarios for a given contaminant/receptor/pathway combination because the RR is a ratio of two ratios, each having the same denominator (either the NOAEL or LOAEL TRV).] These are contaminant/receptor/exposure pathway combinations that represent potential unacceptable risks and that are greater than risks posed by background conditions.

On the basis of results of the QRA, Tables 6–13 and 6–14 summarize the contaminants and media recommended for retention as COPCs for the New Rifle wetland and Roaring Fork pond, respectively, along with the rationale for their retention or elimination. These recommendations will be used in the development of the site compliance strategy, including proposed monitoring requirements. Sediments and plants are grouped together as a medium because it is assumed that plant uptake and concentrations are a function of sediment concentrations. Data were inadequate to evaluate risks posed by intake of other contaminated biota (e.g., frogs) by the great blue heron. However, because concentrations in these biota are dependent on surface water concentrations, a compliance strategy that improves surface water quality will also result in reduced risks to receptors with significant dietary intakes of aquatic organisms.

Table 6–13. Recommendations on COPCs for the New Rifle Wetland (NRW)

Analyte	Medium	Retain as COPC?	Rationale
Ammonia	surface water	ND	data insufficient to fully evaluate
	sediments/plants	N	screened out in statistical analysis
Arsenic	surface water	N	risks not significantly elevated over background
	sediments/plants	N	not detected in plants; risks not significantly elevated over background for sediment
Cadmium	surface water	N	screened out in statistical analysis
	sediments/plants	Y	risks elevated for BI, CC
Fluoride	surface water	Y	risks elevated for BI, CC
	sediments/plants	N	screened out in statistical analysis
Iron	surface water	N	calculated risks are low
	sediments/plants	N	screened out in statistical analysis
Manganese	surface water	N	screened out in statistical analysis
	sediments/plants	N	risks not significantly elevated over background
Molybdenum	surface water	N	calculated risks are low
	sediments/plants	ND	data insufficient to evaluate sediment risks
Nitrate	surface water	ND	data insufficient to fully evaluate
	sediments/plants	ND	data insufficient to fully evaluate sediment; not measured in plants
Phosphate	surface water	N	screened out in statistical analysis
	sediments/plants	ND	data insufficient to evaluate sediment; not measured in plants
Selenium	surface water	N	screened out in statistical analysis
	sediments/plants	N	screened out in statistical analysis in sediment; only 1 detect in plants
Strontium	surface water	N	risks low and not significantly elevated over background
	sediments/plants	ND	data insufficient to evaluate sediment; dietary risks from plants not significantly elevated over background
Sulfate	surface water	N	screened out from statistical analysis
	sediments/plants	N	sediment screened out in statistical analysis; not measured in plants
Uranium	surface water	Y	elevated risks for BI, CC
	sediments/plants	N	screened out from statistical analysis
Vanadium	surface water	N	risks low and not significantly elevated over background
	sediments/plants	N	risks essentially same as background
Zinc	surface water	N	screened out from statistical analysis
	sediments/plants	N	plants screened out from statistical analysis; sediment risks low

ND = not determined BI = benthic invertebrates CC = creek chub

Table 6–14. Recommendations on COPCs for the Roaring Fork Pond (RFP)

Analyte	Medium	Retain as COPC?	Rationale
Ammonia	surface water	Y	risks elevated for BI, CC
	sediments/plants	ND	data insufficient to fully evaluate sediments; not measured in plants
Arsenic	surface water	N	risks not significantly elevated over background
	sediments/plants	N	risks not significantly elevated over background
Cadmium	surface water	N	screened out in statistical analysis
	sediments/plants	Y	risks elevated for BI, CC
Fluoride	surface water	N	risks not significantly elevated over background
	sediments/plants	N	screened out in statistical analysis
Iron	surface water	N	screened out in statistical analysis
	sediments/plants	N	risks not significantly elevated over background
Manganese	surface water	N	screened out in statistical analysis
	sediments/plants	N	risks not significantly elevated over background
Molybdenum	surface water	Y	risks elevated for BI, CC
	sediments/plants	ND	data insufficient to evaluated sediment risks
Nitrate	surface water	Y	risks elevated for MR, MD, GBH
	sediments/plants	ND	data insufficient to fully evaluate sediment; not measured in plants
Phosphate	surface water	N	screened out in statistical analysis
	sediments/plants	ND	data insufficient to evaluate sediment; not measured in plants
Selenium	surface water	N	all risks are low
	sediments/plants	N	only 1 detect in sediment
Strontium	surface water	N	risks low
	sediments/plants	N	screened out in statistical analysis
Sulfate	surface water	Y	risks elevated for most receptors
	sediments/plants	N	risks not significantly elevated above background for sediment; not measured in plants
Uranium	surface water	Y	elevated risks for BI, CC
	sediments/plants	N	not significantly elevated above background
Vanadium	surface water	N	risks low and not significantly elevated over background
	sediments/plants	N	risks essentially same as background
Zinc	surface water	N	screened out from statistical analysis
	sediments/plants	N	sediment risks not significantly above background; predicted plant concentrations low

BI = benthic invertebrates
CC = creek chub
MR = muskrat
MD = mule deer
GBH = great blue heron
ND = not determined

Retention of a contaminant and pathway as a COPC based on calculated HQs and HIs does not necessarily mean that it represents an unacceptable risk, but indicates that it requires evaluation beyond calculation of numerical risk to arrive at an appropriate and informed risk management decision. Potential exposure measurements provide only one indication that adverse ecological effects could be expected at a site. Generally, other lines of evidence are required to support or refute these data (EPA 1997). This evidence could include simple physical observations such as stressed vegetation or fish kills or could involve more extensive surveys of biologic diversity and other ecological parameters. To date, lines of evidence for the evaluation of ecological risk at the New Rifle site have been limited to cursory field observations. No readily apparent effects of ecological degradation have been noted. The following observations are based on the results of the QRA:

- The majority of risks posed by the New Rifle wetland are below or only marginally above acceptable levels. For HQs and HIs exceeding 1, most are only slightly higher than background.

- Risks posed by the Roaring Fork pond are higher than those associated with the New Rifle wetland. The highest relative risks are posed by ammonia, molybdenum, nitrate, and to a much lesser degree, sulfate, in surface water. Risks associated with sediments and dietary intakes are generally low, and potential unacceptable risks (HI or HQ > 1) are only marginally above background. Because the dietary pathway risks are overestimated by using dry weight concentrations, this pathway is considered an insignificant contributor to overall risks.

- Estimated risks to the great blue heron and aquatic plants may be lower than actual risks because of the lack of toxicity data and appropriate uptake factors.

- Population risks (based on LOAEL-based risk calculations) are generally quite low for both New Rifle wetland and Roaring Fork pond locations with the exception of sulfate in surface water at the Roaring Fork pond.

- Because of the highly conservative nature of the QRA, most (if not all) risks calculated probably represent overestimates (exceptions may be the great blue heron and aquatic plants). As such, the New Rifle wetland and Roaring Fork pond probably pose a limited threat to ecological receptors in the area.

6.2.3 Comparison of New Rifle Water Quality with Colorado Agricultural Standards

Table 6–15 presents the mean concentrations of New Rifle wetland and Roaring Fork pond surface water and on-site and downgradient alluvial ground water for New Rifle along with Colorado Agricultural water standards. All constituents in surface water are well below their respective standards with the exception of nitrate in the Roaring Fork pond. Mean nitrate concentrations are essentially at the standard; minimum and maximum concentrations are also very close to the standard as surface water data for this location show only slight variation in nitrate content.

Several constituents in on-site ground water exceed agricultural standards, including fluoride, manganese, selenium, and vanadium. Downgradient ground water also exceeds agricultural standards for manganese; average concentrations for other constituents in downgradient ground water are below standards. It should be noted that maximum manganese concentrations detected in background ground water also exceed the agricultural standard by more than an order of magnitude and that average background concentrations are more than 4 times the standard.

Table 6–15. Comparison of Average New Rifle Surface Water and Alluvial Ground Water Quality with Colorado Agricultural Standards

Contaminant	NRW mean SW mg/L	RFP mean SW mg/L	On-site mean GW mg/L	Off-site mean GW mg/L	Standard[a] mg/L
Ammonia as NH_4	0.18	97.8	145.5	152.3	N/A
Arsenic	0.006	0.0046	0.039	N/A	0.1
Cadmium	0.001	0.0005	N/A	N/A	0.01
Fluoride	0.56	0.45	2.51	1.45	2
Iron	0.01	0.007	1.59	N/A	5
Manganese	0.01	0.16	2.1	2.23	0.2
Molybdenum	0.06	0.5	2.22	0.63	N/A
Nitrate as N	0.136	99	7.7	41	100 (Nitrite & Nitrate as N)
Selenium	0.001	0.0022	0.093	0.006	0.02
Sulfate	131	2754	1298	2586	N/A
Uranium	0.03	0.15	0.11	0.094	N/A
Vanadium	0.01	0.0078	3.17	N/A	0.1
Zinc	0.004	0.0083	n/d	n/d	2

[a]standards based on filtered samples
NRW = New Rifle wetland
RFP = Roaring Fork Pond
N/A = most samples analyzed were below detection
n/d = not analyzed for this constituent
SW = surface water
GW = ground water

6.2.4 Plant Uptake Study

The purpose of this study was to provide information on potential phytotoxic effects of ammonium contaminated ground water encountered in the mitigation wetlands area at the New Rifle site. The study was designed to support both the ERA and revegetation efforts in the mitigation wetlands. Prior to the wetlands reconstruction, elevated levels of both nitrate and ammonium were discovered in ground water at the project site. While high levels of nitrate are generally innocuous, excessive ammonium has been demonstrated to produce toxic effects in plants. However, little information is available on wetland species response to high ammonium level.

Three separate laboratory studies were conducted to evaluate species response at the germination, initial seedling growth, and mature seedling stages. The two treatment factors tested were site water dilution level and pH level. Site water was tested at five separate dilution levels (0%, 10%, 25%, 50%, and 100%) in order to evaluate the response of plant materials to elevated ammonium levels. The pH treatment consisted of three pH levels (6.5, 7.5, and 8.5) per dilution and was utilized to evaluate the response of plant materials to various ammonium levels. All studies were conducted under controlled conditions and randomized block designs. Solutions for all studies were mixed in a similar manner.

The results of the study show that the high levels of ammonium at the site will directly impact vegetation in the mitigation wetlands–site water is toxic to native wetland plant species. Since concentrations vary across the site, however, individual species tolerance may vary. The following conclusions and recommendations are based upon the results of this study:

- Seed germination and initial seedling growth were affected by high ammonium levels as evidenced by reductions in germination percentage, reductions in root and shoot growth, browning of roots and an observed inhibitory effect on algae growth in treatments. In consideration of the revegetation options, levels of 100 parts per million (ppm) ammonium or less within an 18 inch rooting zone should be considered amenable to seeding of tufted hairgrass and alkali grass. At levels greater than 100 ppm ammonium, substantial reductions in shoot growth and stand sustainability can be expected. The upper limits of tolerance for Nebraska sedge are between 40 and 100 ppm ammonium. Planting bullrush or Nebraska sedge by seed is not recommended due to their observed low tolerance to ammonium.

- Mature seedling growth was affected by high ammonium levels in three of five wetland species as evidenced by low shoot biomass levels, chlorotic leaves, leaf loss, drought-like symptoms and moss mortality within containers. Significant mortality was not observed although death appeared imminent for streambank willow and Fremont cottonwood at ammonium levels greater than 100 ppm and 200 ppm, respectively. Nebraska sedge seedlings demonstrated some tolerance with significant reductions only at the 400 ppm NH_4 level. In contrast, elevated levels of ammonium did not affect bullrush and cattail although biomass was substantially reduced in all treatments when compared with the control. In consideration of revegetation options, streambank willow may be planted in substrates exposed to ground water levels up to 100 ppm ammonium; Fremont cottonwood and Nebraska sedge may be planted in substrates exposed to ground water levels up to 200 ppm ammonium; and cattail and bullrush may be planted in substrates exposed to ground water levels up to 400 ppm ammonium.

- Species which reproduce primarily by vegetative means (i.e., bullrush and cattail) should be used in high ammonium areas (> 100 ppm ammonium) in order to avoid inhibition of seed germination.

- Ammonia level as influenced by the pH treatment did not influence results in Investigation 2 or 3. It is assumed that high temperatures (>20 ^{o}C) may have resulted in the volatilization of ammonia, thus preventing fair consideration of this factor. This should be considered in transferring greenhouse scale studies to field conditions.

- Remediation plans that promote nitrification should be developed for areas that will exceed 400 ppm ammonium. Nitrification as a process converts ammonium to nitrate, the non-toxic form of nitrogen. Nitrification is mediated by bacteria and requires substantial amounts of oxygen. Thus, incorporation of organic matter and oxygen into substrates may be necessary to initiate this process.

A Wetlands Management Plan will be developed for the revegetation of the mitigation wetland. Results of this study and any future field studies will be considered in development of this plan.

6.3 Risk Assessment Summary and Recommendations

An evaluation of present-day conditions at the New Rifle site indicates that no risks currently exist for human health. No pathways are present at this time for use of untreated site-related ground water. Wells for the human consumption of ground water have been installed into the alluvial aquifer; however, the wells are required to have treatment systems to ensure that the ground water is safe for drinking. Human health risks presented in this BLRA update are only

potential risks associated with the future use of untreated ground water. The site compliance strategy must ensure that actual site risks are maintained at acceptable levels, both for the 100-year period over which institutional controls are permitted, and the post-100 year time at which the site is assumed to be available for unrestricted access. Human health risk calculations represent worst-case conditions in terms of contaminant concentrations. Potential future risks are based on current levels of contamination; these levels should continue to decrease, as they have from higher concentrations observed in the past. Thus, associated risks will also decline with time.

Ecological risks reported for the New Rifle wetland and the Roaring Fork pond represent a combination of possible present risks and potential future risks. Present risks include those where pathways and receptors currently exist (e.g., ingestion of surface water from the Roaring Fork pond by mule deer and muskrats). Future risks are those that could be present if and when the New Rifle wetland and Roaring Fork pond develop into more viable habitats (e.g., after aquatic plants become established in the Roaring Fork pond and reconstructed New Rifle wetland and are available as a food source for likely receptors). All quantitative ecological risk estimates are somewhat theoretical in that no evidence has been observed to date that indicates site-related contamination has resulted in environmental degradation. In addition, many uncertainties are associated with the quantitative ecological risk calculations, most of which tend to overestimate risks associated with the site.

In developing a compliance strategy that takes into account ecological risks, it must be kept in mind that the majority of risks presented here pertain to potential habitats that have been or are being artificially created. Few of the risks discussed here represent threats to existing healthy populations. In the case of the Roaring Fork pond, it is associated with an ongoing gravel mining operation that is likely to continue for an estimated 10 to 20 years in the future. The disturbed conditions created by the operation are likely to prevent any near-term development of the Roaring Fork pond into an important habitat for wildlife and discourage its use as a significant food or water source for ecological receptors. After gravel mining operations cease in the future and habitat development is more likely, ecological risks will probably have decreased to fairly insignificant levels as natural flushing continues to operate and cause declining contaminant levels.

A large portion of the New Rifle wetland is in the process of being reconstructed. Based on results of the plant uptake study, site-related contamination will probably limit the types of vegetation that can initially thrive in that location. Undoubtedly the same limitations will affect aquatic organisms as well. However, as contamination decreases through time, the New Rifle wetland can be expected to become more diverse and support a greater variety of plant and animal species. Thus, site-related contamination more probably represents a short-term hindrance to wetland reconstruction than a threat to existing viable populations. Risks associated with the already established portion of the New Rifle wetland are generally low, especially in terms of probable population effects (LOAEL-RME scenario, see Table 6–8).

6.3.1 Contaminants of Concern Summary

Table 6–16 summarizes the contaminants that could not be eliminated through application of human health or ecological screening criteria during the risk assessment updates described in the previous sections. These constituents can be considered COCs and must be taken into account in the compliance strategy selection process. Each of these contaminants is discussed below in

terms of their roles in developing an appropriate compliance strategy for the New Rifle site. Discussion of ground water in this section pertains strictly to the surficial aquifer.

Table 6–16. Summary of Human Health and Ecological COCs

Contaminant	Potential Future Human Health Risks		Current/Future Ecological Risks	
	On-site	Off-site	NRW	RFP
Ammonia	Y	Y*	ND	Y
Arsenic	Y	N	N	N
Cadmium	N	N	Y	Y
Fluoride	Y	Y*	Y	N
Manganese	Y	Y	N	N
Molybdenum	Y	Y	ND	ND
Nitrate	Y	Y	ND	Y
Selenium	Y	Y*	N	N
Sulfate	N	N	N	Y
Uranium	Y	Y	Y	Y
Vanadium	Y	Y*	N	N

Y = contaminant is a COC
N = contaminant is not a COC
* = contaminant extends only immediately downgradient of site
ND = not sufficient data to evaluate
NRW = New Rifle wetland
RFP = Roaring Fork pond

6.3.1.1 Ammonia

Ammonia is a COC with respect to both human health and ecological risk. In ground water, elevated concentrations (above risk-based levels for human health) of ammonia extend beyond the site boundary on to the Roaring Fork gravel operation. However, ammonia decreases to acceptable levels in all wells outside the boundaries of the gravel operation.

Based on results of the ecological QRA, ammonia was identified as a COC at the Roaring Fork pond based on surface water concentrations exceeding chronicAWQC. Toxicity data were not available to quantitatively evaluate risks to aquatic plants from surface water or sediment. However, based on results of the plant uptake study, it appears that concentrations of ammonia expected to occur in the reconstructed New Rifle wetland prohibit the growth of many of the aquatic plants used in the study. Very few plant species were able to survive and grow under those conditions.

Natural flushing is expected to result in decreasing ammonia concentrations with time, which will eventually reach levels near background concentrations. However, until that time, institutional controls would be required to prevent improper use of ground water on the New Rifle and Roaring Fork properties. In terms of ecological effects, high ammonia concentrations in surface water would be expected to limit the potential species that can be used to revegetate the New Rifle wetland during near-term reconstruction activities. As ammonia levels decrease, a more diverse plant population could be developed. It would also be expected that ammonia concentrations would inhibit growth of many other aquatic organisms that might otherwise populate the Roaring Fork pond and New Rifle wetland. However, as mentioned previously,

these surface water bodies are artificially constructed and the Roaring Fork pond is still operational. Therefore the effects of ammonia contamination are not destructive, but serve merely to slow the development of a diverse aquatic community. Through natural flushing of the ground water system, contaminant concentrations will decrease, allowing the development of a healthier ecosystem. Ammonia concentrations are expected to decline to acceptable levels within the 100-year period permitted for natural flushing.

6.3.1.2 Arsenic

Arsenic concentrations in ground water exceeding the UMTRA standard of 0.05 mg/L is limited to two on-site wells. Elevated arsenic concentrations do not extend beyond the boundary of the New Rifle site. Arsenic is not a COC for ecological receptors.

Based on comparison with historic data, concentrations of arsenic in ground water have declined by an order of magnitude or more. If these decreases continue, arsenic concentrations should drop to below the UMTRA standard well within the 100-year period allowed for natural flushing. Ground water modeling indicates the MCL could be achieved at the end of a 20-year natural flushing period. Institutional controls would only be required for the New Rifle site itself to prevent exposure to unacceptably high levels of arsenic during the natural flushing period.

6.3.1.3 Cadmium

Cadmium is not a COC for human health and is not of ecological concern in surface water. However, cadmium concentrations in sediment exceed SQC for aquatic organisms at both the New Rifle wetland and Roaring Fork pond. Because SQC are established for the protection of most aquatic organisms, and because conservative assumptions were made in calculating ecological risks (see Section 6.2.2.2), it is likely that only the most sensitive aquatic species would be adversely affected by cadmium in sediments. Because of the operational nature of the Roaring Fork pond and the reconstruction activities associated with the New Rifle wetland, these effects would not cause destruction of any existing populations, but would only serve to prevent the population of those habitats with sensitive species in the future. Measured or predicted concentrations of cadmium in plants that could serve as a food source to other receptors do not result in risks that exceed those posed by background.

Cadmium concentrations in sediment will not be affected by the selection of any given ground water compliance strategy. Because concentrations of cadmium in ground water are below detection at locations likely to recharge the Roaring Fork pond and New Rifle wetland, concentrations of cadmium in sediment should not increase as natural flushing occurs. Decreases of cadmium in sediment could occur through uptake by aquatic plants. Therefore risks to aquatic receptors should decrease through time while not posing significant risks to receptors feeding on those plants.

6.3.1.4 Fluoride

Fluoride is a COC for both human health and ecological risk. Concentrations of fluoride in ground water exceed EPA’s primary drinking water standard in samples from wells from the New Rifle site and from the Roaring Fork gravel site; the agricultural standard for fluoride is also exceeded in these locations. Highest concentrations were detected in wells located at the south central boundary of the Roaring Fork facility near the Colorado River. The highest

concentrations from on-site wells are from those located near the downgradient (western) edge of the site. This distribution of fluoride may be an indication that the plume center has migrated off site to the point where it is discharging to the river; if so, contaminant concentrations would be expected to continually decrease through time. On-site concentrations are approximately half of what they have been historically. Natural flushing should result in concentrations below the MCL well within the 100-year time frame. Institutional controls prohibiting improper ground water use would be required for the New Rifle site and Roaring Fork facility until the time that concentrations have been reduced to acceptable levels.

Fluoride is of ecological concern in surface water at the New Rifle wetland. Concentrations exceed chronic AWQC. This may limit the population of the New Rifle wetland to less sensitive aquatic species in the near term. However, natural flushing should result in fluoride reduction with time to levels that are protective of most aquatic life. Current risks are only marginally above those associated with background and should be reduced to acceptable levels within a relatively short period of time.

6.3.1.5 Manganese

Manganese is not an ecological COC. In ground water, elevated concentrations (above the risk-based level for human health of 1.7 mg/L) of manganese extend beyond the site boundary and on to private property just downgradient of the Roaring Fork gravel operation. However, only wells located on the New Rifle and Roaring Fork sites have concentrations in excess of the range of background concentrations (maximum background concentration for 1998–99 was 3.66 mg/L). Samples from almost all wells (on-site, off-site, and background) exceed the secondary drinking water standard for manganese of 0.05 mg/L; average background is more than an order of magnitude higher than this value. On-site and downgradient wells also exceed the Colorado agricultural standard for manganese; the average background concentration is more than 4 times the standard.

Highest manganese concentrations are from wells located in the southeastern and south central portion of the Roaring Fork property. This distribution of manganese may be an indication that the plume center has migrated off site to the point where it is discharging to the river; if so, contaminant concentrations would be expected to continually decrease through time, at least to within the range of background. Background levels should be reached within the 100-year natural flushing time frame. Institutional controls could be used to prohibit use of ground water with the highest manganese concentrations on the New Rifle and Roaring Fork properties. However, because even background concentrations can exceed the RBC and the agricultural standard, and nearly all locations exceed secondary drinking water criteria (established largely based considerations of taste and odor), some form of water treatment for manganese would probably be desirable even after site-related contamination has diminished.

6.3.1.6 Molybdenum

Molybdenum is a human health and ecological COC. In ground water, elevated concentrations (above the UMTRA standard of 0.1 mg/L) of molybdenum extend beyond the site boundary and on to private property just downgradient of the Roaring Fork gravel operation. Molybdenum decreases to below the standard in all wells in the vicinity of the West Rifle interstate exchange and in those located farther to the south and west. Highest concentrations are from wells located in the southwest quadrant of the New Rifle site. Concentrations decrease farther off site. Current

maximum molybdenum concentrations are almost double the historically observed maximum, though the present mean is slightly less than the historic mean.

Molybdenum is of concern for ecological receptors at the Roaring Fork pond. Concentrations of molybdenum in surface water at the Roaring Fork pond are expected to result in elevated risks from direct contact with aquatic organisms, as well as increased risks through surface water ingestion (specifically by the muskrat). As described for some other contaminants, risks associated with direct contact do not result in destruction or harm to existing populations, but may limit the viability of different species when the Roaring Fork pond becomes a more viable habitat in the future. Ingestion of surface water under current conditions could affect the present muskrat population; however, the operational nature of the Roaring Fork facility probably discourages use of the Roaring Fork pond as a primary source of water intake. Data for the New Rifle wetland are insufficient for evaluating ecological risk due to exposure to molybdenum.

Molybdenum is expected to decline to levels below the UMTRA standard within the 100-year natural flushing period. Ground water modeling indicates the standard will be reached after a period of about 25 years. Until that time, human health can be protected by imposing restrictions on ground water use for the New Rifle and Roaring Fork properties and on the private property immediately west of the Roaring Fork operation. Ecological risks associated with the Roaring Fork pond are expected to be low throughout the operational life of the facility due to lack of a significant hospitable habitat. Elevated risks to muskrat could exist in the future with the establishment of plants at both the Roaring Fork pond and reconstructed New Rifle wetland; risks associated with currently vegetated portions of the New Rifle wetland are elevated for the dietary pathway of the muskrat. Eventually, risks associated with plant intake should decline as continued uptake of molybdenum from sediment causes sediment concentrations to decrease, resulting in corresponding decreases in plant concentrations.

6.3.1.7 Nitrate

Nitrate is a COC for human health and ecological risk. Nitrate in ground water is elevated above the UMTRA standard of 44 mg/L NO_3 as far as 2 miles downgradient of the New Rifle site. The highest nitrate concentrations were detected in wells near the central portion of the Roaring Fork facility. This distribution may indicate that the peak of the nitrate plume has moved off site and that wells upgradient of the current maximum should continue to steadily decline in concentration. The highest current nitrate concentrations are similar to the historic mean concentration. Concentrations in most wells are significantly below the historic minimum concentration.

Nitrate concentrations in surface water at the Roaring Fork pond are high enough to result in elevated risks to ecological receptors via the surface water ingestion pathway (receptors evaluated for this pathway include muskrat, mule deer, and great blue heron). This pathway is complete, but because of the operational nature of the Roaring Fork facility, potential receptors are probably discouraged from use of the Roaring Fork pond as a regular source of drinking water. The wells with the highest nitrate concentrations are located immediately upgradient of the Roaring Fork pond. It is possible that surface water concentrations in the Roaring Fork pond will increase before they begin to consistently decline. Nitrate concentrations in Roaring Fork pond surface water are very close to the Colorado agricultural standard; however, the water is currently not used for agricultural purposes. Other ecological risks associated with nitrate in

surface water and sediment could not be evaluated for the Roaring Fork pond and New Rifle wetland because of lack of data.

Nitrate concentrations in ground water have declined significantly when compared to historical values. The center of the plume has likely moved off site already and is continuing to move downgradient. Based on historic trends, it is likely that nitrate will decline in the ground water to levels below the UMTRA standard well within the 100-year natural flushing time frame. Ground water modeling indicates that nitrate will decrease to below the MCL within 15 years of natural flushing. Risks to human health can be protected in the interim period by imposing institutional controls on ground water use for properties located above the current ground water plume. This includes the New Rifle and Roaring Fork sites, as well as private properties located up to 2 miles downgradient of the western New Rifle site boundary.

Surface water concentrations in the Roaring Fork pond may result in increased risks to ecological receptors through surface water ingestion. However, these risks are expected to decline after the center of the plume migrates downgradient of the Roaring Fork pond and eventually reaches acceptable levels. Risks for the New Rifle wetland could not be evaluated because of lack of data. However, as with the Roaring Fork pond, any potential risks can be expected to decline with time as nitrate concentrations decrease due to natural flushing.

6.3.1.8 Selenium

Selenium is not an ecological COC based on the QRA. However, selenium concentrations in several on-site wells exceeded the agricultural standard of 0.02 mg/L. Some background wells also contain selenium concentrations in excess of the agricultural standard, though average background is approximately half that value. Selenium concentrations in ground water exceed the UMTRA ground water standard. Highest concentrations of selenium were detected in three on-site wells. Most of the samples from these wells exceeded the selenium standard by an order of magnitude or more. Several downgradient off-site locations have selenium concentrations in excess of the UMTRA standard, but only marginally. Background selenium concentrations range up to 0.03 mg/L; all off-site wells have concentrations below this value and all downgradient off-site concentrations are below the human-health RBC of 0.18 mg/L. Based on these results, the only selenium contamination that is clearly site-related and poses potentially unacceptable risks to human health is confined to the ground water beneath the New Rifle site itself. Institutional controls would be required to prevent improper use of on-site ground water. Through natural flushing, selenium levels should decline to at least background within the 100-year time frame. Ground water modeling indicates that selenium concentrations would be reduced to about 0.05 mg/L after 100 years of natural flushing, meeting the SDWA MCL.

6.3.1.9 Sulfate

Sulfate is not a human health COC because the incremental contribution of site-related activities to total sulfate in ground water is very small compared to naturally occurring levels. Additionally, because of the lack of consensus by the scientific community on toxicological effects of sulfate, actual risks due to sulfate exposure cannot be assessed. The average background concentration of sulfate is greater than 1,000 mg/L, and concentrations more than double this are common in background wells. Sulfate is consistently high across the New Rifle study area; sulfate concentrations range to over 2,000 mg/L as far as 3 miles downgradient from the western site boundary.

Sulfate is an ecological COC for the Roaring Fork pond. Risks associated with sulfate in Roaring Fork pond surface water are elevated for all receptors (except the great blue heron for which data were insufficient to evaluate risk), although risks are only marginally above background for receptors other than benthic invertebrates and fish (represented by creek chub). Concentrations of sulfate in surface water collected at the Roaring Fork pond were at the maximum end of the range detected for on site ground water samples and more than twice the average for on site alluvial ground water. The fact that the Roaring Fork pond is part of an operational facility probably discourages its use as a major source of surface water to receptors such as the mule deer and muskrat. Because the pond is artificially constructed, it was not originally a habitat for aquatic receptors such as fish and benthic invertebrates and site-related activities are not responsible for the destruction of a formerly viable habitat.

Background sulfate levels in the reference area pond are an order of magnitude less than concentrations in the Roaring Fork pond, however they are still several of orders of magnitude above both acute and chronic AWQC. Natural flushing should result in decreases in sulfate in the Roaring Fork pond over time; however, “acceptable” levels based on AWQC will probably never be met because of the naturally high concentrations of sulfate in ground water discharging to the ponds.

6.3.1.10 Uranium

Uranium is a COC for human health and ecological risk. The UMTRA standard for uranium is exceeded in wells as far as 3 miles downgradient from the western site boundary. However, background concentrations as high as 0.06 mg/L were detected during the 1998–99 sampling events, exceeding the UMTRA standard of 0.044 mg/L. Natural flushing should result in a decrease in uranium concentration over time, eventually to background levels. Ground water modeling shows that UMTRA standards should be met in 40 years, assuming a background concentration of 0.038 mg/L. In the interim period, institutional controls restricting improper use of alluvial ground water can prevent potential risks to human health. However, because of naturally high uranium in some locations for background alluvial ground water, it is possible that the UMTRA standard may never be met. It may be necessary to establish an ACL for uranium as part of the compliance strategy for the New Rifle site to accommodate the naturally high levels of uranium observed in background alluvial ground water. Even if the UMTRA ground water standard cannot be achieved through natural flushing, the human-health risked-based level of 0.11 mg/L for drinking water will be met and unrestricted use of ground water will result in no unacceptable human health risks after the 100-year natural flushing period.

Uranium concentrations in surface water present potential risks to benthic invertebrates and fish at both the Roaring Fork pond and New Rifle wetland. As noted above for other contaminants, because the Roaring Fork pond is artificially created, site-related contamination will only serve to inhibit habitat development until contaminants have decreased to acceptable levels through natural flushing; no destruction of existing habitat has occurred. Most of the New Rifle wetland is currently undergoing revegetation and does not represent an ecologically significant habitat at this time. Natural flushing of contaminants from the New Rifle wetland as revegetation of the wetland progresses will eventually result in conditions that promote growth and diversification of ecological populations.

6.3.1.11 Vanadium

Vanadium is not an ecological COC. No drinking water standard has been established for vanadium. However, alluvial ground water does exceed the risk-based level of 0.33 mg/L for human health based on regular use as drinking water; it also exceeds the Colorado agricultural standard of 0.1 mg/L. Elevated vanadium concentrations are confined to on-site wells and one immediately downgradient well on the Roaring Fork property (RFN–590). Vanadium in all other wells is at or below detection. Results of ground water modeling indicate that on-site ground water would not meet the risk-based level in 100 years. However, vanadium has displayed significant fluctuations over time, particularly in the highest concentration area. Data are insufficient at this time to understand what factors influence vanadium concentrations and further study is recommended. Institutional controls preventing improper use of ground water during that period will ensure protection of human health; these controls should also restrict the use of on-site ground water for agricultural purposes. Because no drinking water standard exists for vanadium, an ACL will probably need to be established as part of the compliance strategy for the New Rifle site.

End of current text

7.0 Ground Water Compliance Strategy

The framework defined in the final PEIS for the UMTRA Ground Water Project (DOE 1996b) governs selection of the final strategy to achieve compliance with the EPA ground water protection standards. Stakeholder review and acceptance of the final PEIS is documented and supported by the Record of Decision. Presented below is a discussion of how the selection process was used to determine the final ground water compliance strategy at the New Rifle site and a proposed future ground water sampling and analysis plan to monitor compliance and the effectiveness of the selected remedy. Note that the proposed strategy and monitoring focuses on protection of human health. Ecological concerns will be addressed in the *New Rifle Wetlands Monitoring Plan.*

7.1 Compliance Strategy Selection Process

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

- ***No remediation*****—**Compliance with the EPA ground water protection standards would be met without altering the ground water or cleaning it up in any way. This strategy could be applied 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 2.1.1.

- ***Natural flushing*****—** Compliance with the EPA ground water protection 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 if ground water compliance can be achieved within 100 years or less, where effective monitoring and institutional controls can be maintained, and where the ground water is not, and is not projected to be, a source for a public water system.

- ***Active ground water remediation***— Compliance with the EPA ground water protection 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.

7.2 New Rifle Compliance Strategy

DOE is required by the PEIS to follow the ground water compliance selection framework summarized in Figure 7–1 in selecting the appropriate compliance strategy(ies) to clean up areas of the alluvial aquifer affected by former processing activities at the New Rifle site. The lower Wasatch aquifer is not contaminated at the New Rifle site and is therefore not considered in the development of a compliance strategy.

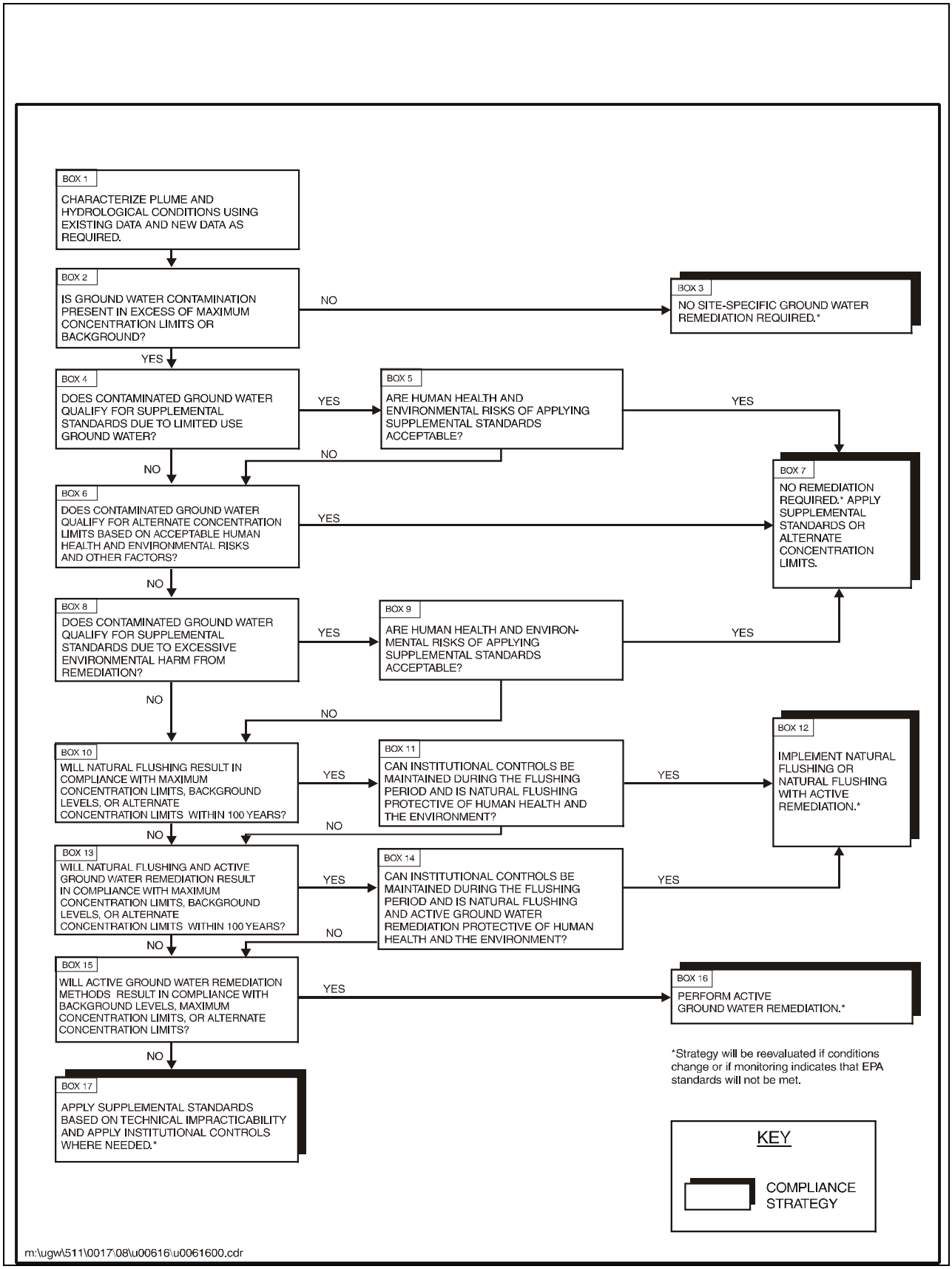

Figure 7–1. Summary of Ground Water Compliance Selection Framework

DOE has determined that natural flushing of the alluvial aquifer, in conjunction with the establishment of ACLs and institutional controls, is the appropriate compliance strategy for remediation of all contaminants at the New Rifle site, except vanadium. Additional data collection is required before an appropriate strategy can be implemented for vanadium. The compliance strategy focuses on COCs retained after completion of the updated human health and ecological risk assessment screening processes (Sections 6.1 and 6.2). An explanation of how the targeted strategy was selected is presented in Table 7–1.

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

Box (Figure 7–1)	Action or Question	Result or Decision
1	Characterize plume and hydrological conditions.	See conceptual site model presented in Section 5.0 and contaminant screening presented in Section 6.0. Move to Box 2.
2	Is ground water contamination present in excess of UMTRA MCLs or background?	Selenium, arsenic, molybdenum, nitrate and uranium exceed the UMTRA MCLs at one or more monitoring points. Ammonium, fluoride, manganese, and vanadium are elevated compared to background and exceed RBCs. Move to Box 4.
4	Does contaminated ground water qualify for supplemental standards due to limited use ground water?	Alluvial ground water is a potential source of drinking water and therefore is not classified as limited use. Move to Box 6.
6	Does contaminated ground water qualify for ACLs based on acceptable human health and environmental risk and other factors?	Concentrations are above proposed ACLs at this time. Move to Box 8.
8	Does contaminated ground water qualify for supplemental standards due to excessive environmental harm from remediation?	Although the applicability has not been formally assessed, it is unlikely that remedial action would cause excessive harm to the environment. Move to Box 10.
10	Will natural flushing result in compliance with UMTRA MCLs, background, or ACLs within 100 years?	Ground water modeling shows that natural flushing will reduce all constituents except vanadium to background, ACLs or MCLs within the 100-year time frame. Data for vanadium are insufficient to evaluate vanadium behavior. Move to Box 11.
11	Can institutional controls be maintained during the flushing period and is the compliance strategy protective of human health and the environment?	The final compliance strategy is protective of human health and the environment. Institutional controls will be in place soon and will prevent use of water. Ground water can be used without restriction after 100 years and will be protective of human health and the environment at that time. Move to Box 12 – implement natural flushing.
12		Implement natural flushing .

Each of the compliance strategy components is discussed in the following sections.

7.2.1 Natural Flushing

The 100-year time period allowed for natural flushing will commence upon finalizing the GCAP and concurrence by the NRC. Results of ground water contaminant transport modeling are presented in Section 5.3.5 and Appendix D. Predicted concentrations for arsenic, molybdenum, nitrate, selenium, uranium, and vanadium after 100 years of natural flushing are summarized below.

Modeling results for arsenic using the most recent sampling data suggests that the UMTRA standard of 0.05 mg/L could be met in 20 years of natural flushing. This is consistent with

decreases in arsenic concentrations observed to date. Monitoring will take place to track the progress of natural flushing.

Molybdenum is predicted to decrease to levels below the UMTRA standard of 0.1 mg/L after a period of about 25 years. The background concentration used for the modeling was 0.019 mg/L, and none of the background wells exceeded the MCL.

Uranium is predicted to decrease to levels below the UMTRA standard after a period of about 40 years. However, it should be noted that a background concentration of 0.038 mg/L uranium was used for purposes of ground water modeling. This is the average calculated background uranium concentration. Levels of uranium in excess of 0.06 mg/L have been observed in one background well. Therefore, the compliance standard for uranium in site ground water may be either background or the UMTRA MCL. The monitoring strategy is designed to account for variations in background uranium that may exceed the UMTRA standard.

Based on modeling results, maximum selenium concentrations are predicted to decrease to 0.05 mg/L, approximately the SDWA MCL and proposed ACL, in about 100 years. Background wells had concentrations of selenium up to nearly twice the UMTRA standard based on results from the most recent sampling round. However, these concentrations are all below the risk-based level of 0.18 mg/L.

Nitrate is predicted to decrease to levels below the UMTRA standard of 44 mg/L after 10 years of natural flushing. Modeling did not take into account geochemical and biological reactions that could affect nitrate behavior. However, a conservative modeling approach was used and it is anticipated that nitrate will be below the UMTRA standard well within the 100-year natural flushing period.

Modeling was not performed for ammonia, fluoride, and manganese, as no UMTRA standards exist for these contaminants. However, based on historical trends, it is likely that these constituents will reach background levels or other applicable standards within the 100-year natural flushing time frame. Monitoring will be conducted for these constituents to ensure that contaminant trends are consistent with this assumption and that acceptable levels can be reached.

7.2.2 Vanadium Pilot Study

No drinking water standard exists for vanadium; however, plume concentrations currently exceed RBC for human health as well as background levels and phytotoxicity values for plants. Modeling predictions indicate that lowering concentrations of vanadium to the 0.33 mg/L RBC (the proposed ACL) will require about 300 years of natural flushing. However, these results are not consistent with recently observed decreases in vanadium from approximately 25 mg/L in 1998 to 12 mg/L in 1999 for well RFN–658. Historically, plume values for vanadium were much lower than this (less than 2 mg/L); the observed high values were not observed until surface remediation activities were conducted. Because of the uncertainty surrounding the behavior of vanadium, it is proposed that a pilot study be conducted to further investigate the nature of vanadium contamination and to monitor its behavior over time. Preliminary indications are that active remediation would be required to reduce concentrations to the RBC, but data are insufficient at this time to select that option as the appropriate compliance strategy (particularly because no UMTRA standard exists for vanadium). Upon completion of the pilot study, the selected option for vanadium will be one of the following: 1) natural flushing, 2) soil removal,

3) in-situ stabilization, or 4) pump and treat. Options 2 through 4 are discussed further in Section 8 of this document. Documentation of the remedial decision for vanadium will be included in the GCAP.

7.2.3 Alternate Concentration Limits

Because vanadium exceeds the background concentration and has no drinking water standard, an ACL is required. A concentration of 0.33 mg/L is proposed as the ACL for vanadium. This is the human health RBC for use of water in a residential setting. The volume of ground water exceeding the proposed ACL of 0.33 mg/L is relatively small. After evaluation of vanadium behavior is completed during the pilot study, a remediation strategy protective of human health and the environment will be proposed.

An ACL is also proposed for selenium of 0.05 mg/L. This is the SDWA MCL. An ACL is warranted because background concentrations of selenium exceed the UMTRA standard. All observed concentrations of selenium are below the RBC of 0.18 mg/L and are protective of human health.

Because no standard exists for ammonia, it is also necessary to establish an ACL for that constituent. Based on site-specific conditions, an ACL of 200 mg/L (measured as NH_4) is proposed. This is approximately the risk-based level based on 0.28 mg/L as NH_3 and an average site pH and temperature.

An application for these ACLs will be submitted to the NRC for their concurrence.

7.2.4 Institutional Controls

An institutional control program is being developed to prevent the future use of the potentially harmful contaminated ground water associated with the New Rifle site during the 100-year natural flushing period. The institutional control mechanisms will include the use of both zone district changes and deed restrictions.

- The zone district changes will prohibit the use of the untreated contaminated shallow ground water for drinking purposes for humans. The zone changes will apply to the entire contaminated area plus a buffer area. Efforts are ongoing with the City of Rifle and Garfield County to initiate the zone changes. It is anticipated that the zone changes will be effective by the spring of 2000.

- Because the ground water contaminant concentrations are increased and additional contaminants occur in the ground water beneath the New Rifle site and the UMETCO, Inc. property (Roaring Fork gravel operation), located immediately to the west of the New Rifle site, more stringent restrictions will be placed on these properties to prohibit the use of the untreated contaminated ground water for drinking water, livestock watering, irrigation of edible vegetation, and ponding uses for ornamental or landscaping purposes. However, the current sand and gravel mining activities at the UMETCO property will not be affected by the deed restriction. In order that a deed restriction is legal and enforceable, it must be initiated at the time of a land ownership transfer. Efforts are underway to establish deed restrictions at these two properties. It is anticipated that both deed restrictions will be in place by the spring of 2000.

The New Rifle Site Institutions Controls Program, once finalized, will be effective and legally enforceable over the 100-year natural flushing period. If contaminants flush more quickly than predicted, restrictions on ground water use may be lifted sooner on a location by location basis.

7.3 Monitoring Compliance Strategy

7.3.1 Monitoring Requirements for the Alluvial Aquifer

The monitoring strategy for the alluvial aquifer is designed to determine the progress of the natural flushing process in meeting compliance standards for site COCs. Standards for Se, V, and ammonia are their proposed ACLs of 0.05 mg/L, 0.33 mg/L, and 200 mg/L (as NH_4), respectively. For uranium, the cleanup goal is the UMTRA ground water standard of 0.044 mg/L or background, whichever is higher. For arsenic, the cleanup goal is the UMTRA ground water standard of 0.05 mg/L, and for molybdenum the cleanup goal is the UMTRA ground water standard of 0.1 mg/L. The cleanup goal for nitrate is the UMTRA standard of 44 mg/L and the goal for fluoride is the SDWA MCL of 4 mg/L. The cleanup goal for manganese is approximately background, or 4 mg/L. Monitoring will focus on these contaminants.

In 1998, 23 permanent alluvial wells were installed at the New Rifle site. Included in this installation were several wells that were intended to serve as background wells and to define the extent of contamination at the site. Specifically, well RFN–172 was installed about 3 miles downgradient of the site to determine the longitudinal extent of contamination, and wells RFN–169 and –173 were installed upgradient to the east and north, respectively, to serve as background. Sampling at these locations in August 1998 and January 1999 showed that the water at each exceeded the UMTRA standard for one COC: uranium for the downgradient well RFN–172 and selenium for the upgradient wells RFN–169 and –173. However, concentrations of uranium detected most recently at location RFN–172 are within the range of background. The only wells in the current sampling program which do not exceed the standards for any of the New Rifle COCs are well RFN–171, which is not as far downgradient as well RFN–172 but is further north, on the north side of Interstate 70; well RFN–215, located on the east side of the site along the southern edge of the former ore storage area; and well RFN–220, located north of the railroad track between wells RFN–172 and –620. Well RFN–172 may represent the approximate leading edge of the plume. Uranium was not detected in the most downgradient location at temporary well RFN–186. Uranium contamination does not extend as far to the east as well RFN–215, nor as far to the north as wells RFN–171 and –173.

Wells RFN–218, –657, –658, and –659 have been established as appropriate for monitoring the “hot spots” with highest concentrations of vanadium, arsenic, and selenium, as well as elevated concentrations of most other constituents. Based on the ground water modeling, these contaminants are likely to take the longest time to flush from the system. In addition to the “hot spot” wells denoted above, wells RFN–169, –170, –171, –172, –173, –195, –201, –210, –213, –215, –216, –220, –590, and –635 have been established as appropriate for monitoring progress of the natural flushing in the alluvial aquifer. The perimeter of the plume is approximately defined by well RFN–172 to the southwest; wells RFN–171, –170, –173, and –213 along the northern boundary; well RFN–169 to the northeast; and wells RFN–215 and –216 to the southeast. In no case is the concentration of any COC at any of these perimeter wells more than twice the standard, whereas some of the other wells exceed treatment standards by a factor of five or higher for many COCs. Background wells RFN–169 and –173 will be used to monitor

background concentrations of uranium to assist in establishing the appropriate cleanup standard (UMTRA standard or background, whichever is higher).

The other wells in the natural flushing monitoring program are intended to monitor the progress of the flushing in key points within the boundaries of the plume. Well RFN–220 is north of the railroad track between wells RFN–172 and –620, and currently does not show contamination for any of the New Rifle COCs. Well RFN–210, which exceeds the UMTRA standard for uranium only (but is within the range of background), is along the railroad tracks near the Colorado River south of Interstate 70. Wells RFN–195 and –201, which exceed the cleanup goals for ammonia, nitrate, molybdenum and uranium, are located to the west and east, respectively, of the Roaring Fork gravel pit, approximately on the north-south centerline of the current plume.
Well RFN–590 is immediately south of the borrow pit pond and exceeds UMTRA standards for molybdenum, nitrate, selenium, and uranium, and cleanup goals for ammonia, manganese, and vanadium. Well RFN–635 is well to the south of the borrow pit pond, south of Interstate 70 along the Colorado River; it exceeds UMTRA standards for molybdenum, nitrate, selenium, uranium and cleanup goals for ammonia, fluoride, and manganese. The proposed monitoring locations are shown on Figure 7–2.

Results of ground water modeling for six of the COCs at the New Rifle site (arsenic, molybdenum, selenium, uranium, nitrate, and vanadium) are presented in Section 5.3.5. This modeling indicates that concentrations of all of these COCs are expected to decrease and that the contaminant plumes for all these COCs are not expected to spread beyond their current boundaries. The monitoring plan is intended to confirm that the natural flushing is progressing approximately in conformance with the predictions of the model, that constituents that were not modeled decrease as well, and that no contaminant spreading is taking place.

Sampling of each well will take place annually for 10 years. If a COC sampled in a well is below standards for 3 consecutive years, cleanup will be considered complete at that location and monitoring will be discontinued for that COC. After the first 10 years of monitoring have been completed, monitoring will be reduced to every 5 years until the year 2030, assuming COCs continue to show a decline through the first 10-year period. At that time the monitoring strategy will be reevaluated and adjusted as appropriate based on previous results. Any COC that is below standards will be analyzed for 3 consecutive years to eliminate it from further analysis. If the monitoring of wells at the perimeter of the contaminant plumes shows that contaminants have begun to spread beyond the current plume boundaries, or if some other changes in contaminant trends are noted, the sampling plan may also be reevaluated and adjusted at that time.

Monitor wells not required as part of the monitor network will be abandoned according to applicable State of Colorado regulations and UMTRA project procedures. Abandonment will be accomplished in the near future under the LTSM program.

Monitoring requirements for surface water, if necessary, will be addressed in the Wetlands Management Plan for the mitigation wetland.

Table7– *2. Summary of Monitoring Requirements*

Location	Monitoring Purpose	Analytes	Frequency[a]
RFN–218, –657, –658, –659	Monitor hot-spots	As, V, Se, Mo, U, NH_4, NO_3, Mn, F	Annually for 10 years; every 5 years thereafter until 2030
RFN–170, –210, –172	Monitor middle and leading edge of Mo and U plumes.	Mo, U	Annually for 10 years; every 5 years thereafter until 2030
RFN–169, –173	Monitor background to establish appropriate standard for uranium; ensure no upgradient spread of plumes	As, V, Se, Mo, U, NH_4, NO_3, Mn, F	Annually for 10 years; every 5 years thereafter until 2030
RFN–171, –220, –213	Monitor spread of plume to uncontaminated areas downgradient	Mo, U	Annually for 10 years; every 5 years thereafter until 2030
RFN–215, –216, –201, –195, –590, –635	Monitor flushing in main body of plumes	As, V, Se, Mo, U, NH_4, NO_3, Mn, F	Annually for 10 years; every 5 years thereafter until 2030

[a]Monitoring for a COC will be discontinued if below standards for three consecutive sampling rounds.

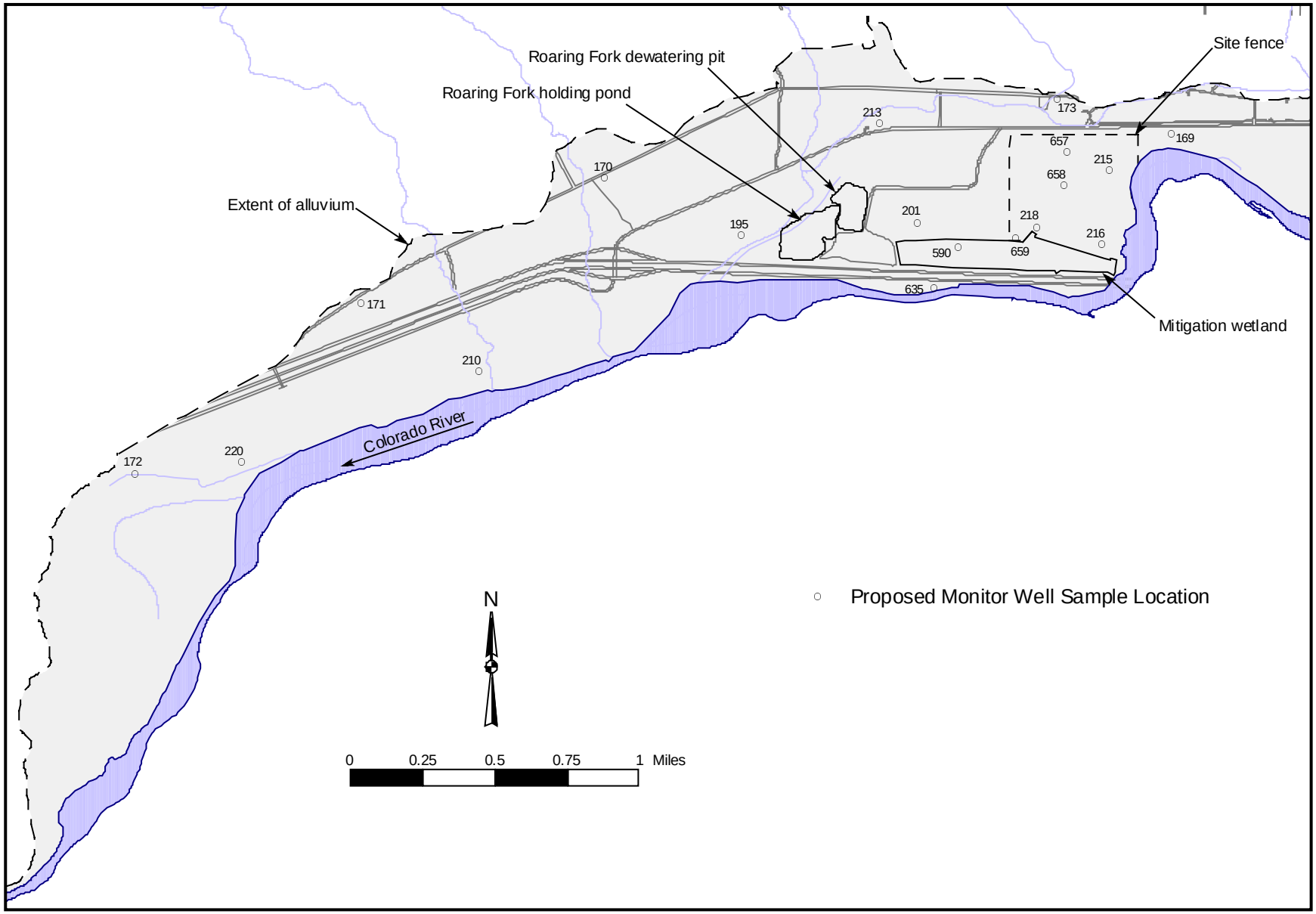

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Figure 7–2. Proposed Monitoring Locations for the New Rifle Site

8.0 Development and Evaluation of Active Remediation Alternatives

As presented in Section 7.0, the selected alluvial ground water compliance strategy for the New Rifle site is natural flushing possibly supplemented by active remediation for vanadium to alleviate hot spots. The purpose of this section is to develop and evaluate different active remediation alternatives and recommend an alternative for possible remediation of the vanadium contamination, should active remediation be necessary.

Section 8.1 gives an overview of the process used to evaluate and screen technologies and alternatives, including a detailed explanation of the evaluation criteria. Section 8.2 develops a list of potential technologies that could be used for remediation of the site, evaluates the technologies, and screens out technologies that are not feasible. Section 8.3 lists technologies that passed the initial screening, combines the technologies into alternatives, and develops the parameters that will be used for the detailed evaluation, which is presented in Section 8.4. Section 8.5 presents the proposed alternative for active remediation, along with discussions of how the proposed method may be deployed and of the uncertainties and limitations of the proposed alternative.

8.1 Process for Development and Evaluation of Technologies and Alternatives

This section gives an overview of the process used to arrive at a proposed alternative for remediation of vanadium contamination in ground water at the New Rifle site. It also includes a description of the criteria used to evaluate technologies and alternatives.

8.1.1 Overview of the Process

The general process used to select a proposed alternative for remediation of contamination at an UMTRA ground water site is:

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

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

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

8.1.2 Evaluation Criteria

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

8.1.2.1 Effectiveness

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

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

Remediation Time Frame

The remediation time frame is dependent on the volume of contaminated ground water present at the site, and the rate at which contaminated ground water can be extracted from the aquifer. DOE has established 20 years as a general goal for active remediation of contaminated ground water for the UMTRA Ground Water Program. However, the New Rifle remediation is planned as a “hot spot” cleanup of the most heavily contaminated area of the plume, and the volume of this contaminated water is much lower than the volumes of contamination at most other UMTRA ground water sites. The estimated area of the New Rifle “hot spot” is about 1,000 ft square, with a saturated thickness of 15 ft. Assuming a porosity of 0.25, this gives a pore volume of 3.75 million cubic ft or 28,052,000 gallons. Treating this volume of water over a period of six months, with an on-stream (overall reliability) factor of 90 percent, would require a flow rate of 120 gpm. Because of the high K_d of vanadium, it is unlikely that the vanadium extraction can be completed with pumping of a single pore volume of water. For comparison of treatment processes, it will be assumed that a complete extraction will require treatment of three pore volumes, over an appropriate time period.

Conformance with Aquifer-Restoration Standards and Goals

The general requirements for contaminant levels in ground water at UMTRA sites are specified as MCLs in 40 CFR 192.04, Table 1. The average concentrations of the various COCs, in the

background water and in the alluvial water in the area of the site, are shown in Table 8–1. Also shown are the MCLs for each constituent. The value listed for vanadium is a risk-based ACL rather than an MCL, since no MCL has been established for vanadium.

Table 8–1. Contaminant Concentrations in Alluvial Ground Water at New Rifle Site

Constituent	Average Background	On-Site	MCL or ACL Standard
Ammonia (as NH_4)	0.1358	196.9	200[a]
Arsenic	ND	0.02943	0.05
Fluoride	0.8409	2.358	4.0
Molybdenum	0.01893	1.998	0.1
Nitrate	4.025	69.26	44
Selenium	0.01092	0.07212	0.01 [a]
Uranium	0.03719	0.1149	0.044
Vanadium	ND[b]	2.412	0.33 [a]

[a]ACL
[b]ND = not detected

Natural flushing will reduce the concentrations of these contaminants, with the exception of vanadium, to below their respective cleanup goal within the statutory time limit of 100 years. Vanadium, as explained in Section 5.3.5.1, cannot be removed to the levels called for by the ACL by natural flushing within the 100-year time limit, and may require active remediation to reduce the level enough that the standard can be met by natural flushing.

The maximum concentration of vanadium in the alluvial aquifer is due to a localized “hot spot,” located around well 658. Vanadium concentrations outside this hot spot are expected to flush naturally. The extraction process will be concentrated in the “hot spot” area. The concentration of contaminants at well 658, and at the other wells where vanadium concentrations exceed the RBC standard of 0.33 mg/L, is shown in Table 8–2. All data in this table is based on averages of the two most recent sampling data sets, collected in August 1998 and January 1999.

Table 8–2. Contaminant Concentrations (mg/L) in Vanadium Plume

Constituent	Well 658	Well 659	Well 657	Well 590	Plume Average
Ammonia (as NH_4)	325	222	25	648	305
Arsenic	0.193	0.074	0.0022	0.0004	0.0671
Cadmium	0.0088	0.0137	0.0009	0.0177	0.0102
Fluoride	3.47	4.92	0.68	3.40	3.12
Molybdenum	5.59	5.92	0.21	2.46	3.54
Nitrate	3.6	244	14.7	72	83
Selenium	0.472	0.107	0.192	0.0196	0.198
Sulfate	2,485	2,220	604	4,815	2,531
Uranium	0.230	0.111	0.0493	0.0791	0.117
Vanadium	19.1	4.54	1.07	0.423	6.28

Short-Term Effects

Short-term effects are a consideration of the potential effects to the community, workers, and the environment. The New Rifle site is located adjacent to the Colorado River and near the community of Rifle, Colorado. The nearest residence is located within 1 mile of the site. Given

the nature of the contaminants and the treatment processes being considered, it is unlikely that persons living in the area would be affected by the operation of the treatment facility. However, selection of treatment processes should keep in mind the potential for major upsets.

The site lies between U.S. Highway 6, which passes immediately to the north of the site, and Interstate 70, which passes to the south on the south side of the Colorado River. Both of these highways are major thoroughfares carrying large volumes of through and local traffic. Treatment processes will have to be evaluated to determine whether upsets in operation could conceivably result in emissions that could affect travelers on these highways.

Evaluating the effects to workers entails considering the risks to persons employed to construct the treatment system and to those employed to operate and maintain the system during its operational life, as well as persons supporting the remedial action, such as samplers and equipment operators disposing of treatment residuals.

The evaluation of short-term effects also considers environmental effects. Environmental effects include potential environmental harm caused by deployment of a technology or alternative and whether the potential harm of remediation outweighs the benefits to be derived from restoration of the alluvial aquifer.

Disposal of Treatment Residuals

Active treatment processes can produce a significant amount of residual waste. This may include dissolved solids from the ground water, as well as the residuals from any other chemicals that may have been added during the treatment process (e.g., antiscalants or softening agents). These residuals must be contained during the remediation process, and disposed of either during or at the end of remediation. The need to keep residuals contained during remediation to minimize their volume, and to dispose of them as needed, may become a significant cost over the lifetime of the remediation process.

8.1.2.2 Implementability

Implementability is an assessment of the feasibility of building, operating, and maintaining a remediation system.

The following aspects of feasibility will be discussed in this SOWP:

- Ease of construction.
- Ease of operation and maintenance.
- Expected reliability.
- Ability to handle changes in influent composition.
- Ability to handle increases in extraction capacity.

Construction

Skilled construction labor is expected to be reasonably available in the immediate vicinity of the New Rifle site, and utilities are readily available. Thus the evaluation will place no premium, other than that of direct construction cost, on treatment systems which are easier to construct. Consideration of construction also requires examining the uncertainty associated with construction, such as the potential for schedule delays caused by technical problems.

Expected Reliability

Reliability is defined as the probability that a system will meet required performance standards. This includes both the physical reliability of the equipment comprising the system, and the process reliability, which considers the potential for variability in process performance both on a day-to-day and on a year-to-year basis. Evaluation of the potential reliability of a treatment system must consider the technical and operational complexity and required level of training for operators.

Ability to Handle Changes in Influent Composition

The treatment system which will be implemented at New Rifle will be intended to remediate vanadium. Remediation of other constituents is not a requirement of the remediation program. However, some technologies are less constituent-specific than others are, and changes in concentrations of other contaminants may affect the functioning of the treatment process. Some technologies are better suited to handle such variations than others are, and this ability will be considered in evaluating whether specific technologies are suited for use at the New Rifle site. The possibility also exists that the natural processes that are being relied upon for the bulk of the remediation may prove less effective than the current modeling suggests, and that active treatment might therefore be required to reduce concentrations of one or more COCs. However, given the short duration of the vanadium treatment compared with the duration of the natural flushing process, the possibility of having to treat for one or more additional contaminants is not expected to be realized within the operating lifetime of the vanadium treatment process. Therefore, it will not be considered in this evaluation.

Ability to Handle Increases in Extraction Capacity

The volume of the contaminated plume in this SOWP is a conservative estimate based on sampling from a number of wells at the site. The likelihood that the actual volume of the contaminant plume will eventually be found to be significantly higher than the present estimate is not considered high, but the possibility must be allowed for. The ability of a remediation system to handle such increases, and the incremental cost involved in doing so, must be considered in evaluating whether specific technologies are suited for use at the New Rifle site.

8.1.2.3 Cost

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

The cost evaluation considers both the direct capital cost and the total cost of treatment over the life of the project, determined by combining the initial capital cost for the treatment system with the estimated O&M costs over the project duration, using a net present worth analysis. By discounting all costs to a common base year, the costs for expenditures in different years can be compared on the basis of a single figure (i.e., the net present worth). Guidance issued by the Office of Management and Budget (OMB) was used to calculate net present worth. The guidance recommends using a real interest rate (i.e., a rate that does not consider inflation) to discount out-year costs that have not been adjusted for inflation.

Where possible, direct capital costs are developed from invoice costs of similar systems. If that information is not available, generic unit costs, vendor information, and conventional cost-estimating guides have been used. O&M costs are based on labor costs, energy costs, material and equipment costs, and maintenance costs.

8.2 Evaluation of Technologies

8.2.1 Extraction Technologies

8.2.1.1 Conventional Vertical Wells

Because of the shallow depth and flat terrain of the alluvium aquifer at the New Rifle site, the only type of extraction-well system to be considered is conventional vertical wells. Vertical wells are the most commonly used ground water extraction devices, so the bulk of field experience and knowledge relates to conventional vertical wells. Installation of vertical wells is relatively straightforward in most cases and, when combined with proper well design, construction, and development, vertical wells can provide very satisfactory yields. Vertical extraction wells can be readily converted to injection wells as needed, or vice versa, and can also be easily decommissioned when necessary. Finally, the theoretical performance of a vertical well can be simulated analytically or numerically during the design process using readily available and accepted mathematical formulations, while no comparable knowledge base exists for other technologies. Thus, vertical wells are the only extraction technology recommended for detailed evaluation.

Simple extraction, however, is inadequate for the requirements of the New Rifle site. Vanadium, the principal COC at New Rifle, has a high K_d and so cannot be removed to the treatment level by extraction of as many as 50 pore volumes. Two strategies could be used to remediate the vanadium:

(1) Ground water chemistry could be altered to favor vanadium partitioning into the liquid phase, which could then be removed by pumping.

(2) Sorptive materials could be added to increase the amount of vanadium partitioned into the aquifer solids, which would stabilize the vanadium in situ.

The remainder of this section will consider these alternative approaches.

8.2.1.2 Vanadium Mobilization

Much of the cost involved in pump-and-treat remediation is attributed to the large volume of ground water that often must be removed and treated to remove the contaminants that are bound to the solid fraction. By decreasing partitioning into the solid phases, vanadium could be flushed out of the formation with less pumping and at a lower cost.

Much of the vanadium in ground water is in the +5 oxidation state called vanadate. Vanadate occurs as an anion in solution and readily complexes with hydroxyl ions causing it to be more sorptive at low pH values and more mobile at elevated pH values. Thus, increasing pH using bases such as sodium hydroxide could enhance flushing. Increasing the pH from 6.8 (approximately the pH of the Rifle ground water) to 10 will increase partitioning into the soluble phase by several orders of magnitude.

Alternatively, adding a chelating or complexing agent to the ground water could decrease the distribution ratio. An example is the use of citrate, which is known to chelate with iron. Because iron tends to stabilize vanadate, the addition of citrate should mobilize vanadate. Other agents are probably known from wastewater treatment, ore milling, and ground water remediation industries that could be used to enhance the flushing of vanadium.

A literature search and treatability study, consisting of both laboratory and field testing, are recommended prior to pilot testing of any technology to enhance the mobility of vanadium into the liquid phase. The testing would have to demonstrate both the technology itself, and the injection process that would be required to inject the lixiviant and assure adequate subterranean mixing.

8.2.1.3 In-Situ Stabilization

As an alternative to enhancing the mobility of the vanadium into the liquid phase, the vanadium could be stabilized in place by increasing the amount partitioned into the immobile solid fraction. If the sorbent concentration in the aquifer is increased, partitioning of vanadium to the immobile solids will be enhanced and the concentration in ground water will be reduced.

One means of increasing the sorptive portion of the aquifer is to introduce ferric oxyhydroxide. This can be accomplished by injecting dissolved ferric chloride into the aquifer. The acidic ferric chloride solution reacts with alkaline aquifer materials and precipitates ferric oxyhydroxide which immobilizes vanadium by incorporating it in a semicrystalline structure. Treatability tests and a pilot demonstration would be required to determine the acid-neutralizing capacity of the aquifer and the ability of the ferric oxyhydroxide to incorporate vanadium, as well as the feasibility of the injection process.

8.2.2 Effluent Discharge Technologies

This section describes the various ways in which treated water from the treatment process can be discharged.

8.2.2.1 Evaporation

Evaporation treats extracted ground water by allowing the water to evaporate under conditions in which the nonvolatile contaminants are contained and allowed to concentrate for later disposal. It may also be used to treat concentrated waste water from treatment processes that produce a waste-water stream. The advantages and disadvantages of evaporation as a treatment option are discussed in the "Evaporation" article in Section 8.2.3.

8.2.2.2 Discharge to Wetlands

In this option, the treated water would be used to irrigate the wetlands located immediately south of the site between the site and the Colorado River. The most likely method would be to construct an infiltration trench immediately upgradient of the wetlands. The feasibility of this option would depend on the quality of the treated water, since discharge to the wetlands would require a permit from the Colorado State Department of Environmental Quality. Since the ammonia concentration in the vanadium plume is in the toxicity range for wetland plants, this discharge option could only be used if the treatment process included ammonia reduction. The expected ammonia concentration is around 300 mg/L, and plant toxicity appears to become an issue above about 100 mg/L, so the ammonia (as NH_4) would have to be reduced by about two-thirds for this discharge option to be viable.

8.2.2.3 Discharge to Surface Water

Under this option, the extracted and treated ground water would be discharged directly to the Colorado River at a rate of about 100 gpm or about 162 acre-ft per year. The feasibility of this option would depend on the quality of the treated water, since discharge to the Colorado River would require a permit from the Colorado Department of Public Health and Environment.

In 1993, Roaring Fork Resources was granted a permit by the Colorado Department of Health to discharge up to 2,000 gpm of water from its UMETCO pit located adjacent to the New Rifle site. The 30-day averages and daily maximums from that permit are shown in Table 8–3.

Table 8–3. Effluent Limitations for Roaring Fork Resources

Effluent Parameter	Discharge Limitations	
	30-day Average	Daily Maximum
Ammonia, mg/L as N	51	N/A
Nitrate, mg/L as N	14	N/A
TDS, ton/day	Report	1.0
Uranium, mg/L	2.0	4.0
Arsenic, mg/L	0.5	1.0
Zinc, mg/L	0.5	1.0
pH, max-min	N/A	6.5 - 9.0

N/A = not available

Assuming that the effluent limitations of the Roaring Fork Resources permit are expected to be similar to what would be required for effluent discharge from the New Rifle site, the major contaminants that would require treatment before the effluent could be discharged to the Colorado River are ammonia and TDS. The Roaring Fork Resources limit of 1.0 ton TDS per day is equivalent to only 83 mg/L at their permitted maximum flow rate of 2,000 gpm. Even if

the same 1.0 ton per day standard were applied to the New Rifle remediation, the effluent TDS concentration would have to be less than 1,400 mg/L. This would not be possible without substantial reduction in the sulfate concentration. Therefore, treatment technologies that do not reduce sulfate could not be used if the preferred discharge process is direct discharge to the Colorado River.

8.2.2.4 Injection Wells

With this option, injection wells would be used to conduct the treated effluent directly back into the alluvial aquifer. The use of injection wells inside the plume area can contain the spread of contaminants, and may also boost the pumping rate that can be realized because injection into the plume surcharges the hydraulic heads in the pumping zone, potentially allowing a higher rate of extraction.

Injection wells would be designed in accordance with specifications attributed to extraction wells, and considerable care would be required for all aspects of well completion. With injection wells, the suspended sediment concentration in particular would need to be very low to help prevent clogging. Thus, the feasibility of this option could depend on the treatment process selected, since some treatment processes might not improve suspended solids concentrations. Other factors to consider with injection wells are the consequences of air entrainment and the entrance velocities for the treated effluent.

8.2.2.5 Discharge to Municipal Water Treatment Facility

Under this option, the effluent from the New Rifle treatment unit would be discharged to the Rifle municipal water treatment plant, which is adjacent to the site. Obviously this would require that the effluent from the treatment process be acceptable to the authorities responsible for the operation of that plant. This would mean, at a minimum, that the treatment process should remove radionuclides and heavy metals. The necessity for removal of ammonia, vanadium, or other contaminants would also have to be established before this option could be selected.

8.2.2.6 Sale to Municipal Water Supply

If the quality of the treated water is sufficiently high (e.g., if a distillation treatment process is used), it may be possible to sell the water to the Rifle municipal water supply system. The quantity would be only about 115 gpm, or a total of about 84 million gallons over the lifetime of the treatment process. However, this could be a viable disposal option if the proper arrangements could be made.

8.2.2.7 Effluent Discharge Technologies Recommended for Detailed Evaluation

The selection of effluent discharge technology depends on a number of factors, such as the quality of the treated water produced, which impacts state permitting requirements; the need of the wetlands for irrigation; and the nature of the treatment technology chosen. The effluent discharge technology cannot be considered by itself, but must be considered as part of the overall remediation process. No effluent discharge technology will be ruled out at this point of the evaluation.

8.2.3 Treatment Technologies

Many treatment processes were identified as potentially applicable for treating the vanadium-contaminated ground water at the New Rifle site. The processes can be categorized as follows:

- Evaporation systems.
- Distillation systems.
- Through-medium processes, such as continuous ion exchange.
- Chemical treatment processes.
- Membrane separation processes, including reverse osmosis (RO) and nanofiltration.

This section will review the potential applicability of these treatment processes and eliminate those that are obviously unsuitable. It will also review the feasibility of removing ammonia as a discrete process, and of removing the contaminated soil for disposal elsewhere. The processes which are not eliminated in this first screening will be evaluated in greater detail in Section 8.3.

8.2.3.1 Evaporation Systems

Solar evaporation consists of putting the water into large lined or unlined outdoor ponds at influent rates that match the rate of natural evaporation. It is an established method for reducing the volume of contaminated surface or ground water, that does not contain volatile hazardous compounds, in arid and semiarid regions of the United States. Nonvolatile contaminants will not evaporate and, instead, will concentrate as a sludge that must be periodically removed for disposal. Solar evaporation systems are constrained by climatic effects, notably temperature (solar radiation), humidity, and wind.

Although the New Rifle site receives only about 11 inches of annual precipitation per year, the relatively cool climate makes it a rather poor candidate for implementation of an evaporation process. Pan evaporation rates at the New Rifle site (Section 4.0) average about 24 inches per year, and precipitation rates equal or exceed evaporation rates for 5 months out of the year. Allowing for the reduction in effective evaporation rates at higher salinity, as the contents of an evaporation pond become more concentrated, the net evaporation rate at the New Rifle site averages less than 5 inches per year. This low evaporation rate means that the surface area required to achieve complete evaporation in a simple solar pond at New Rifle is so large as to render this option impractical; preliminary calculations suggest that solar evaporation of a constant flow of 100 gpm would require a solar evaporation pond having an area of nearly 340 acres.

The effectiveness of solar evaporation systems can be greatly enhanced by adding spray systems in which water is sprayed as a fine mist into the air above the solar pond. The fine mist droplets evaporate much more readily than does the bulk water at the pond surface. Use of a spray system can substantially reduce the size of the pond required. For instance, addition of a spray system could reduce the size of the evaporation pond for the New Rifle site from nearly 340 acres required for a simple solar evaporation pond treating a flow rate of 100 gpm, to about 7 acres for a spray pond treating the same rate, which is still a considerable area. In addition, a spray system

considerably increases the complexity of the system and requires more maintenance and operator attention than simple solar evaporation.

A further consideration in implementing an evaporation system at New Rifle is that evaporation will result in complete loss of the water, as well as total removal of all constituents. In the case of New Rifle, the only constituent for which removal is necessary is vanadium. The total TDS in the alluvial ground water is about 3,400 mg/L. The vanadium concentration is about 13 mg/L, less than 0.5 percent of the total. But an evaporation process will remove all TDS. Thus, 260 pounds of innocuous contaminants will be removed for every pound of vanadium. These solids must be removed and disposed of at the conclusion of the treatment project.

Nonetheless, because of its generally low cost and technical simplicity, spray evaporation remains, in concept at least, an attractive treatment technology for use in any system where it is climatologically feasible and recharge of treated water is not a requirement. Therefore, evaporation with spray enhancement has been retained for detailed evaluation as a treatment alternative.

8.2.3.2 Distillation Systems

In a simple distillation process, water is vaporized and the water vapors are condensed and recovered as clean water. Nonvolatile contaminants such as nitrates, sulfates, uranium, and other components of TDS will not evaporate and will be left behind in the evaporation chamber, where they will concentrate and must be bled off (removed at a slow rate). The condensed water can be reinjected into the aquifer, discharged to surface water, or used for any other purpose that would be appropriate. The concentrate, or brine, may be taken off site for disposal; alternately, it may be evaporated to dryness in a small solar pond or in another process (such as a brine crystallizer), and the residue can then be disposed of as a solid.

Distillation is one of the most expensive treatment technologies to implement, because of the significant capital costs of distillation systems. However, distillation does recover almost all of the water as a product water, which is of very high quality, containing virtually no dissolved or suspended solids. Energy requirements for distillation units can be greatly reduced by the use of “vapor recompression,” in which the heat that is given off by the condensation of the water vapor is recovered in a fan or compressor and used to preheat the feed water. Evaporation of water using a standard boiler with no energy recovery requires almost 2,400 kilowatts per hour (kW-hr) of electricity per 1,000 gallons of water evaporated. Commercial vapor recompression distillation systems can process 1,000 gallons of water while consuming as little as 35 kW-hr. This low energy consumption makes distillation more nearly economically competitive with other treatment processes.

Like evaporation, distillation removes all of the TDS, not just the targeted contaminants. Therefore, byproduct disposal costs may be relatively high. Distillation has already been chosen as the primary treatment technology at the Tuba City and Monument Valley UMTRA sites, and was demonstrated in a pilot study conducted at Tuba City in the fall of 1998. The knowledge gained during that study, which confirmed the applicability of distillation to cleanup of UMTRA ground waters, could be applied directly to the New Rifle site. Cost evaluations for the Tuba City and Monument Valley sites indicated that the low operating costs of distillation systems could offset the higher capital costs over a long project lifetime. Therefore, distillation was selected for detailed evaluation as a treatment alternative.

8.2.3.3 Through-Medium Processes

In a through-medium process, a flow stream is passed through a column or reactor containing an insoluble adsorptive or exchange medium. Synthetic ion exchange resins, which are manufactured to have high affinities for certain types of ions, are widely used in through-medium processes for removal of uranium and many other dissolved ionic contaminants.

Another type of through-medium process uses zero-valent iron (ZVI) to remove a wide varitey of contaminants from ground water. Two field demonstration projects, including one at an UMTRA site, are testing the feasibility of this process for removal of radionuclides and metallic contaminants from ground water. These projects use ZVI in-situ, in the form of Permeable Reactive Treatment (PeRT) walls. The feasibility of implementing a PeRT wall at the New Rifle site is considered problematic because the principal contaminant of concern at New Rifle is vanadium, and the remediation of vanadium will require implementing an in-situ process for mobilizing vanadium. The process for desorbing vanadium from the solid matrix will require addition of a lixiviant, pH adjustment, and/or other means to reduce the Kd of vanadium. (See Section 8.2.1.2.) Adjusting the in-situ conditions to mobilize vanadium is expected to adversely impact the effectiveness of an adjacent PeRT wall.

The effectiveness of ZVI as an ex-situ treatment process is being evaluated in a laboratory study at GJO. An ex-situ process would permit the neutralization of the additive(s) used to mobilize vanadium, prior to introduction of the ground water into the treatment reactor. Although the GJO ZVI study is primarily intended to generate design data for an ex-situ reactor, the results of the study could also be applied to the design and implementation of a PeRT wall if the mobilization process ultimately selected makes use of a PeRT wall possible.

Conventional ion exchange processes are generally impractical for liquids having dissolved solids loadings higher than about 1,500 mg/L, due to high elutriation rates at higher solids levels. However, in the case of the New Rifle site, the anions that make up the bulk of the TDS (by weight) do not need to be removed. The total TDS in the vicinity of well 658, where the extraction well(s) will be placed, averages about 5,000 mg/L, but the cationic species constitute only about 1,500 mg/L of this total. And most cationic ion-exchange resins have at least some measure of selectivity; they will remove some cations but not all of them. This is helpful in minimizing waste generation, because all species that are removed in the ion-exchange process must subsequently be recovered for disposal in the regeneration process.

Meeting the recommended treatment level of 0.33 mg/L of vanadium will require a removal efficiency of about 96 percent, assuming that the influent composition is approximately as shown in Table 8–2. Commercial ion-exchange resins are available which will remove both vanadium and ammonia, as well as uranium, molybdenum, and lead. The attractiveness of a process that could remove these species without affecting the nitrate level is obvious. Testing of ZVI has indicated that it can achieve the required level of efficiency at very short residence times. It will also remove arsenic and uranium at high efficiency. But its efficiency for ammonia removal has not been demonstrated. Therefore, ZVI would not be a promising candidate as a stand-alone treatment process for New Rifle unless laboratory studies can demonstrate its effectiveness for ammonia removal. However, ammonia removal is not technically difficult (see “Ammonia Removal,” following.)

Ion exchange and ZVI may be viewed as competitors because both operate in a flow-through mode. The advantages of ion exchange include its higher state of development, because commercial ion-exchange systems have been in service for many years; and the fact that it can serve as a stand-alone process for vanadium and ammonia removal. Advantages of ZVI are its demonstrated effectiveness at removal of heavy metals, and its low cost. And it cannot be stated at this time that ZVI cannot remove ammonia; at the current state of process knowledge, all that can be said is that the effectiveness of ZVI for ammonia removal has not been demonstrated. Considering all of the relevant factors, the ZVI process, combined with an appropriate process for removal of ammonia, was chosen as the through-medium process for detailed evaluation and treatment alternative for the New Rifle site.

8.2.3.4 Chemical Treatment Processes

Chemical treatment systems using precipitation, coagulation and flocculation, gravity settling, and filtration processes, generally include addition of chemicals for pH adjustment, formation of precipitates, and the like. Such systems are widely used for treatment of contaminated waters produced during remediation of former uranium mill sites. They are very effective for removal of COCs such as uranium, radium, and sulfate. However, conventional chemical treatment processes are not effective for removal of vanadium, because vanadium does not form insoluble compounds with sulfate, nitrate, chloride, or other readily available anions, and insoluble vanadium species such as vanadium carbide and simple chemical processes cannot form vanadium nitride. Nor can ammonia readily be precipitated in a chemical process for the same reason; relatively insoluble ammonium species such as ammonium magnesium phosphate and ammonium manganese phosphate cannot be formed without addition of large amounts of expensive reagents. Therefore, chemical treatment is not appropriate for use in the New Rifle remediation, and was not retained for detailed evaluation.

8.2.3.5 Membrane Separation Processes

Membrane separation includes all processes in which extremely fine or molecular-level filters are employed. The fine filter, operated under pressure, allows clean water to pass through the element as a clean stream, or permeate, on the downstream side of the element, while the contaminants collect as a concentrate stream, or brine, on the upstream side. The most commonly-employed membrane separation processes, in increasing order of effectiveness in removal of dissolved ionic species, are ultra-filtration, nanofiltration, and RO. As a general rule, the more completely a membrane separation process removes contaminants from an aqueous stream, the more brine is produced.

Membrane-separation processes are, generally speaking, non-specific; that is, they reject a very high percentage of nearly all common contaminating species in ground water. An RO water treatment system was used at the Monticello, Utah, Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA) site for 2 years for removal of selenium, uranium, and radium from extraction water generated during remediation of the tailings pile at that site. The RO system effectively removed not only these species, but almost all other ions from the water, resulting in an effluent that could be directly discharged to the adjacent creek with no other treatment required. In this sense, RO is not too different from distillation and evaporation, two other technologies which would remove not only the vanadium, but virtually all other contaminants, from the treated water, resulting in significantly increased waste generation and subsequent disposal costs.

However, even RO, which has the highest filtration efficiency of any of the membrane separation processes, is virtually ineffective for removal of ammonia. Thus, a separate treatment process would be required to remove ammonia. Also, compared to distillation, RO generates relatively large quantities of waste brine, which must be handled and disposed of. The capital costs of RO systems are relatively high, if less so than, for example, distillation. Finally, DOE's experience with the Monticello RO unit indicates that RO units require more operating and maintenance manpower than other processes such as distillation.

For use at the New Rifle site, then, RO, or other membrane-separation processes, combines ineffectiveness for removal of ammonia, generation of large amounts of byproduct waste, and high demands for skilled operating manpower, while offering no offsetting advantages for the remediation program. Therefore, membrane separation was not selected for detailed evaluation as a treatment alternative.

8.2.3.6 Ammonia Removal

Removal of ammonia can be achieved relatively simply by adjusting the pH of the water to favor the ammonia form, which is volatile and can be flashed off, over the ammonium ion, which is nonvolatile and will remain in the liquid. The equilibrium distribution is

$$NH_3 + H_2O \; NH_4^+ + OH^-$$

The equilibrium constant, K, is 1.75×10^{-5}. So at a moderately basic condition of pH = 10, the NH_4^+/NH_3 ratio is 0.175:1, and about 85 percent of the ammonia would exist as ammonia and would flash off as gas. But high ammonia removal can be achieved at lower pH by adding residence time. This allows any volatile ammonia in the water to be stripped off into the vapor phase. As ammonia is stripped off, a small amount of the ammonium will be converted to ammonia to maintain the equilibrium. This ammonia will be stripped off in turn, and further ammonium will be converted. The result in this case is removal of over 800 times more ammonia than the equilibrium predicts.

Implementation of an ammonia-stripping process at New Rifle would require a source of caustic addition for pH adjustment, and a circulating column or large open, agitated tank to allow flashing of the volatile ammonia. Bench testing would be required to determine the optimum combination of pH, for low chemical consumption, and residence time, for small tank or column size. Since ammonia stripping is a low-cost and low-impact process, which is required for successful implementation of other processes that do not affect ammonia, it was retained for detailed evaluation.

8.2.3.7 Soil Removal

In this alternative, the contaminated soil in the vanadium hot spot would be removed from the site and disposed of at a licensed landfill. The excavated soil would be replaced with compatible noncontaminated soil. Extraction water produced during the excavation operation would be treated to remove ammonia and vanadium.

The attraction of soil removal is a relatively high certainty of success. Pretreatment of the ground water to mobilize vanadium, which is the most problematic aspect of pump-and-treat water

treatment processes at New Rifle, is not a consideration for soil removal because the source of the vanadium is removed. Treatment of the excavation water would be straightforward. However, the large quantity of soil that would have to be removed at New Rifle renders the process impractical. Assuming that the quantity of saturated soil that would have to be removed is 1,000 ft square with a depth of 7.5 ft, the volume of contaminated soil is 277,778 cubic yards. This is a worst-case estimate based on the current extent of the vanadium plume. An estimated cost for removal of this material and disposal at the nearest available landfill, the Cheney repository outside Grand Junction, Colorado, is over $8 million. This does not include the cost of treating the excavation water. And the available capacity of the Cheney repository is only about 250,000 cubic yards, some of which is earmarked for other programs. A redesign of the repository could be required to increase its capacity to accommodate the additional material from New Rifle.

Therefore, soil removal must be rejected at this time as impractical. However, at the present time, there is limited information available on the extent of the vanadium contamination, if any. Additional characterization will be performed at the site to more accurately assess the volume of vanadium-contaminated soil. Should this characterization establish that the actual volume is substantially less than the 277,778 cubic yards used in this estimate, the practicality of soil removal as a remediation process may be investigated further.

8.3 Evaluation of Alternatives

This section combines technologies evaluated in the previous section into pumping alternatives and treatment alternatives. The pumping alternatives make use of the extraction and disposal technologies retained for detailed evaluation. The treatment alternatives make use of the treatment technologies retained for detailed evaluation.

8.3.1 Pumping Alternatives

8.3.1.1 Vertical Pumping Wells

As described in Section 8.2.1. the only pumping alternative selected for the New Rifle site is conventional vertical wells. This being the case, there is no need for an evaluation of effectiveness or implementability. Pumping tests performed at the New Rifle site in 1998 established that wells in the alluvial aquifer could sustain flows of at least 15 gpm, and in some cases as high as 80 to 90 gpm. The desired flow rate to the treatment system is 120 gpm, and in order to maximize the effectiveness of the extraction process, the extraction wells should be spread over as much of the plume area as possible. Therefore, the cost estimate assumes that a total of nine extraction wells will be installed. These should easily supply at least the required 120 gpm with one or more wells out of service at any one time.

The total capital cost for this pumping alternative, including all nine wells, pumps, controls, and piping to direct the water from the well field to the treatment system, is estimated at $0.21 million. Considering the small number of wells required for the extraction system, and the fact that the New Rifle site is located only 1 hour from the GJO, it is assumed that the operation of the extraction system will not require operators on site. Any operating manpower required for the extraction system will be supplied by the operator(s) assigned to the treatment system. Therefore, annual O&M costs, for power and maintenance materials, are estimated at $6,200. The net present value for this pumping alternative is estimated at $0.221 million.

8.3.1.2 Vanadium Mobilization

As described in Section 8.2.1, the process for mobilizing vanadium from the solid phase so that it can be extracted by the extraction wells has not yet been developed and demonstrated. Obviously, the process must be defined before it can be costed. However, the vanadium mobilization process is common to all three treatment processes, so for the purposes of comparing the costs of the three systems, the capital and operating costs of the pretreatment process can be neglected.

8.3.1.3 In-Situ Stabilization

Like the mobilization process, the process for stabilizing vanadium in situ has not been developed or demonstrated, so no meaningful cost estimate can be prepared at this time. The cost of in-situ stabilization will probably be similar to the cost of vanadium mobilization, since both processes consist of addition of some chemical and development of a process for injecting the chemical into the ground in such a way that subterranean mixing is optimized. But in-situ stabilization will not required extraction, treatment, or effluent disposal systems. So it seems likely that, if in-situ stabilization can be demonstrated, it will be by far the least expensive alternative.

8.3.2 Treatment Alternatives

The treatment alternatives evaluated in this section are:

- Treatment Alternative 1—Evaporation with Spray Enhancement
- Treatment Alternative 2—Distillation
- Treatment Alternative 3—Zero-Valent Iron with Ammonia Stripping

All treatment alternatives will use conventional vertical wells for removal of the contaminated ground water. The disposal of the treated water will depend on the treatment technology selected. Spray evaporation does not produce treated water. Distillation produces very high-quality treated water that is suitable for disposal by any of the options described in Section 8.2.2. The quality of the treated water produced by ZVI will not be as high as that produced by the distillation system, since ZVI will not significantly affect the concentrations of constituents such as sulfate which constitute the bulk of the TDS. The effluent from ZVI would not be suitable for discharge to surface water or for sale to a municipal water supply.

The treatment systems will be designed to treat 3 pore volumes, over an active life of 6 months per pore volume. The spray evaporation system must operate seasonally, during the 6 months out of the year that evaporation exceeds precipitation at the New Rifle site, and therefore treatment of three pore volumes using spray evaporation can be expected to take 3 years. Distillation and ZVI can operate year-round, so treatment of three pore volumes will take 18 months. As described in “Remediation Time Frame” in Section 8.1.2 above, the estimated quantity of water in one pore volume of vanadium-contaminated water that must be treated is 28,000,000 gallons (28 million gallons), so the total volume of water to be treated is 84,000,000 gallons (84 million gallons). Treatment of one pore volume in 6 months, assuming operation of the extraction and

treatment system for 365 days per year, 24 hours per day, with a 90 percent on-stream factor, will require a flow capacity of 120 gpm. All cost estimates for the extraction, treatment, and injection systems shall be designed for this capacity. The costs of the treatment processes, which utilize vanadium mobilization, will be computed without the added cost of the vanadium mobilization process.

Cost estimates for all treatment processes will be compared based on a net present worth, calculated over the total project life, using the OMB standard discount rate of 7 percent.

No cost basis will be developed for the in-situ stabilization process, and it will not be compared directly with the other processes. If in-situ stabilization is technically feasible and acceptable to all stakeholders and regulatory bodies, it will be at least as effective as any of the treatment processes and should be less costly. In-situ stabilization will not be addressed in the comparative evaluation of the treatment alternatives that follows.

8.3.2.1 Treatment Alternative 1—Evaporation with Spray Enhancement

Effectiveness

The evaporation rate for a spray nozzle designed for continuing operation under high solids loading levels is about 5 to 10 percent water loss per pass through the nozzle (Bete Fog Nozzle 1994). The addition of a spray system that employs 324 nozzles, each with a flow capacity of 40 gpm (hence, an evaporation rate of 2 gpm, or 5 percent of the total flow), increases the evaporation capability by as much as 628 gpm at the height of the solar evaporation season, and enables the required pond size to be reduced to around 3 acres.

A disadvantage of a spray system, as opposed to a simple evaporation system, is that the water in the simple evaporation pond may be evaporated to dryness, so only a single large pond is required. A spray system, however, can only be operated as long as the pond contents remain liquid. Once the liquid in the pond reaches a certain concentration of solids, the efficiency of the spray system begins to drop dramatically. The concentration of solids at this point is still low enough that disposal is impractical without further concentration. The sludge mixture must be evaporated further by solar evaporation before disposal. This can be done either by taking the spray pond out of service and letting the water evaporate in the pond or by transferring the sludge to a second pond for dewatering. Given the short operating lifespan of the New Rifle pond, the capital cost of a sludge pond cannot be justified. The spray pond can be converted to a simple solar evaporation pond for the sludge at the end of the project.

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

Evaporation meets the requirements of 40 CFR 192 and is protective of human health and the environment. The only residual produced is the concentrated sludge.

Implementability

Addition of a spray system to an evaporation pond adds complexity and requires a significantly higher degree of oversight than a simple solar evaporation system. But the low annual

evaporation rate at New Rifle renders this consideration moot, since a simple solar evaporation pond would require an impracticably large surface area; a spray system is the only feasible way to implement a solar evaporation option at New Rifle.

The spray system could operate continuously, although the rate of evaporation from the sprays would be reduced at night. The New Rifle system would spray water that is contaminated with radionuclides into the atmosphere. The initial concentration of radionuclides in the water is low but would increase significantly as the pond contents became more concentrated. Operating such a system without continuous monitoring has the potential for loss of radionuclide containment. A pond designed for continuous operation of the spray system would be smaller and less costly to construct than a pond sized to operate only during daylight hours or during a day shift. The cost of staffing for continuous operation must be weighed against the additional capital cost for building a larger pond. Preliminary estimates indicate that the larger pond would pay for itself in reduced O&M costs in 2 years. Given the low mobility of the vanadium, it is unlikely that the treatment can be completed in less than 2 years. Therefore, the system will be designed assuming operation only during the day shift.

The system design parameters are:

- Influent rate of 120 gpm.
- Spray system operates 12 hours per day, for approximately 6 months a year (April through September).
- System reliability of 85 percent. This allows for equipment breakdowns and nozzle plugging, which would affect parts of the system at different times, and shutdown of the entire system during high-wind periods.
- Evaporation rates varying from 5 percent of total flow during periods of low solar evaporation to 10 percent of total flow during periods of high solar evaporation.
- Maximum pond depth of 3 ft.

With these design conditions, the required double-lined pond area is approximately 3 acres, and about 324 nozzles are required.

Operating the system will require the following principal functions: embankment inspection and maintenance, liner inspection and repair, water-level monitoring, circulation pump monitoring and maintenance, spray system monitoring and maintenance, and monitoring for leaks. A single operator can perform all these functions during the day shift. Installing and maintaining adequate fencing to keep livestock and wildlife away from the pond can minimize the need for inspections.

Monitoring for leaks will consist primarily of monitoring the water levels in the sumps of the leak detection system. This can be done remotely using a telemetry system. Leak detection pump status can also be monitored remotely using telemetry. Maintenance and repair of pumps and spray nozzles is an on-site function, but round-the-clock presence of maintenance personnel is not required because the spray system will not be operated continuously.

The principal environmental compliance issue associated with maintaining large, lined ponds is uncontrolled release through overflow or leaks. Use of double-lined ponds and an interliner leak detection system will control subsurface releases. Such engineering controls are highly reliable. Overflow of the large pond used in this treatment system is unlikely because the water level changes relatively slowly due to the large size of the pond, and because the pond will be monitored on a regular basis by operating personnel.

A large, open body of water in an arid region attracts birds and insects, creating a potential exposure pathway for contamination. Over time, the concentration of uranium, metals, and metalloids (e.g., selenium) in the pond water will increase. Birds and insects may be attracted to the ponds and exposed to high levels of contaminants. The risk increases with a spray system in which contaminants become airborne. Thus, the ability to control waterfowl and insect access to heavily contaminated water will be a concern.

Waste disposal will not be an ongoing function for the evaporation system because the bulk of the concentrated sludge can be disposed of at the end of the remedial action. Final disposal will entail stabilizing and removing about 2,300 tons of sludge from the ponds and transporting the mass to an authorized disposal site. The pond liner system will also be removed and disposed of at an authorized disposal site at the end of its service life.

Cost

The capital cost of the 120-gpm spray evaporation system, including the direct capital cost of building the new pond and spray system, and all indirect costs associated with design, such as subcontractor monitoring, is estimated at $1.20 million. The annual O&M cost of the spray system, including utilities, operator time, and expected maintenance, is estimated at $0.37 million. The estimated present worth value of treatment alternative 1, projected over the total estimated treatment time of 3 years, is $2.18 million dollars. Costs do not include those required for sludge and liner removal and disposal.

8.3.2.2 Treatment Alternative 2—Distillation

Effectiveness

Evaporation and water recovery using simple distillation is an established and proven technology for treatment of contaminated water. A distillation unit will consistently produce a product effluent containing less than 50 mg/L of dissolved solids, and will often meet or exceed drinking water standards with no further treatment required. The concentrated “brine,” which contains essentially all of the dissolved solids, radionuclides, and other nonvolatile contaminants from the original feed, typically averages 5 percent or less of the total feed, depending on the concentration of contaminants in the feed.

Table 8–4 shows data that was developed during pilot testing at the Tuba City site of a distillation system similar to that which would be used at New Rifle. All concentrations are given in mg/L.

Table 8–4. Performance of Distillation Unit

Parameter	Influent	Effluent
Sulfate	2,440	0.824
Nitrate	819	2.48
Ammonium	61.9	2.09
TDS	4,900	37
Uranium	0.146	<0.001

The overall level of TDS in the contaminated water that will be treated at New Rifle will be about 3,500 mg/L, which is similar to the Tuba City water from which the data above was taken. Based on these data, the likelihood that the treated effluent from the distillation system will be able to meet or exceed the applicable treatment standards is extremely high.

Pretreatment for the feed water is expected to consist of addition of sulfuric acid for removal of carbonate, and an antiscalant to minimize fouling of the heat-transfer surfaces.

Distillation meets the requirements of 40 CFR 192 and is protective of human health and the environment. The treated effluent is of high quality, while the volume of the concentrated brine is less than that produced by the other active processes.

Implementability

Commercial distillation units are self-contained and include all instrumentation required for monitoring and controlling the operation. The units are designed for outdoor operation with no building required. The operation of the unit can be monitored at a remote location using the instrumentation and computer software provided as part of the package. The electricity demand of the distillation unit is low.

Commercial distillation systems are reliable and generally require a low level of oversight and only scheduled maintenance during their operating life. Installation of the distillation unit will be straightforward, and can be done by project construction personnel. Operation of the distillation system will require a minimum of managerial and technical supervision. The acid pretreatment system can operate unattended, although periodic replenishing of the acid will be required, as well as occasional maintenance. The cost estimate for the operation of the distillation system includes two full-time employees for operation and maintenance.

The distillation process continuously generates concentrated brine. The concentration of solids in the brine discharged from the distillation unit is low enough that disposal is impractical without further concentration. The brine must be evaporated further, perhaps to dryness, by dewatering via solar evaporation. The size of a solar evaporation pond to evaporate 6 gpm of concentrated brine is estimated at almost 11 acres, which is impracticably large for the New Rifle site. Therefore, a spray evaporation system, similar to that employed in Treatment Option 1 except smaller in size, will be used. A double-lined spray evaporation pond will be constructed for this purpose. For a discussion of the implementability of spray evaporation ponds, see the “Implementability” section under “Treatment Alternative 1—Evaporation with Spray Enhancement,” above.

Commercial distillation units are modular in design. Increasing the capacity of the overall system above the current design capacity will require addition of more distillation units unless additional capacity is specified as a design requirement.

Cost

The capital cost of the distillation system, including the evaporation pond and required ancillary equipment, is estimated at $3.63 million. Most of this cost is for procurement of the treatment system itself, which is estimated to cost $2.28 million. Much of this cost could be recovered if the distillation treatment system could be reutilized at another site after the treatment work at New Rifle is completed. Annual operating costs will be about $0.425 million. Since this treatment system will operate for only 18 months, the present worth cost of this treatment alternative, projected over the total treatment time, is $4.41 million.

8.3.2.3 Treatment Alternative 3—Zero-Valent Iron with Ammonia Stripping

Effectiveness

During the spring of 1998, a number of bench-scale studies were performed, using ground water from the Monticello, Utah, (CERCLA site) to measure the effectiveness of ZVI for removal of the contaminants in the water at that site. Table 8–5 gives the measured removal efficiency of the ZVI for a number of different constituents, at superficial residence times of 6 and 40 minutes, as well as the expected concentrations of the constituents in the effluent from a ZVI reactor. These effluent concentrations assume that the composition of the influent can be established by averaging the composition of well RFN–658 with the average of the other wells in the vanadium "hot spot", as show in Table 8–2.

Table 8–5. Performance of ZVI

Constituent	Influent Composition mg/L	MCL mg/L	6 Minutes Residence Time		40 Minutes Residence Time	
			Removal Efficiency, %	Expected Effluent Conc.	Removal Efficiency, %	Expected Effluent Conc.
Arsenic	0.130	0.05	98	0.003	98	0.003
Molybdenum	4.57	0.1	31	3.15	67	1.51
Selenium	0.335	0.01	53	0.157	96	.013
Uranium	0.174	0.044	>99.8	<0.001	>99.8	<0.001
Vanadium	12.7	0.33	>99.7	<0.038	>99.7	<0.038

Based on these data, the effluent from a ZVI reactor will be well below the MCLs for arsenic, uranium, and vanadium, and well above the MCL for molybdenum. The selenium concentration is difficult to predict. The Monticello treatability studies established that the effectiveness of ZVI for selenium removal depends on whether the selenium is present as the selenite, Se^{+4}, or as the selenate, Se^{+6}. ZVI removes selenite quickly and essentially totally, but the removal of selenate is much slower. The effectiveness of ZVI for removing selenium from the New Rifle ground water would have to be established by laboratory testing.

Nitrate removal was evaluated in a different study. This was a longer-term study, which compared the performance of various ZVIs not only at different residence times, but also at longer life times. The results of this study are given in Table 8–6. Life time was measured in

reactor volumes, that is, total effluent volumes equivalent to the volume of the reactor itself. The results show that nitrate removal was affected more by the time that the reactor had been in service than by the residence time.

The only known data on ammonia removal with ZVI was collected as part of the Durango, Colorado, permeable barrier project. These data, consisting of three data points from two different cells, showed significant discrepancies in removal efficiency, with one point showing near total removal, and another showing a fivefold *increase* in ammonia concentration. No residence times were available. Influent ammonia concentrations were less than 1 mg/L, so the relevance of this data to the New Rifle site is debatable.

Table 8–6. Nitrate Removal Using ZVI

Cumulative Reactor Volumes	Residence Time, minutes	NO_3 Concentration in Influent mg/L	NO_3 Concentration in Effluent mg/L
9.6	248	18	7.75
46.5	138	17.2	2.87
141.2	71	17.8	7.0
201.5	86	17.5	0.016

The efficiency of ammonia stripping depends on the pH of the solution and the residence time. Bench-scale testing will be needed to optimize the design of the stripping system. The influent to the treatment system is expected to contain about 300 to 350 mg/L of ammonia. The desired ammonia level in the effluent is about 100 mg/L. The system will need to achieve a stripping efficiency of about 65 to 70 percent in order to meet this target. This would correspond to a theoretical pH of about 8.9 for a simple reactor having no residence time. The ground water in the vicinity of well 658 is mildly acid, with a pH of about 6.8, and it is not heavily buffered. So the pH adjustment should not require large amounts of makeup chemicals even if the residence time is short.

Implementability

The ZVI system will be relatively simple to design and construct. The ammonia stripper will consist of a chemical addition system and either a stirred tank or a stripping column. The ZVI process requires only a tank to contain the ZVI, and the residence time required by the ZVI process for vanadium removal is sufficiently short that a relatively small vessel size will be needed.

The great simplicity of the ZVI system should make it very simple to operate. The cost estimates for this system assume that a full-time operator will not be required. Given the proximity of the New Rifle site to the GJO, it is anticipated that a part-time GJO person visiting the site 3 to 4 times per week can handle operator duties for the ZVI system. Travel time to and from the site will probably take more man-hours than on-site time.

The waste streams generated by the ZVI system will be the vent stream from the ammonia stripper, which will consist of ammonia and water vapor and which will be vented directly to the atmosphere; and the spent ZVI, which will be removed and disposed of at the end of the project. Although ZVI removes radionuclides at a high efficiency, the radionuclide concentration in the New Rifle ground water is low enough that the spent ZVI should not require classification as a

residual radioactive material. No data is available to predict the long-term lifespan of the ZVI, but the initial fill of ZVI is expected to last for at least the 18-month duration of the remediation.

Cost

Due to the simplicity of the ZVI and ammonia-stripping processes, the capital cost of the ZVI-ammonia stripping system is estimated at $76,000. Assuming that operating labor is on a part-time basis, as described above, the annual operating costs will be about $57,000. The present worth cost of this treatment alternative, projected over the total estimated time of 18 months, is $0.18 million.

8.4 Comparative Evaluation of Alternatives

The following section compares the three pump-and-treat treatment alternatives and recommends a proposed treatment alternative for implementation at the New Rifle site. The treatment alternatives are compared with one another on the basis of each of the evaluation criteria presented in the introduction to this section. In-situ stabilization of vanadium is not included in this comparison. As described in Section 8.3.2, if in-situ stabilization is technically feasible and acceptable to all stakeholders and regulatory bodies, it will be at least as effective as any of the pump-and-treat processes and should be less costly, and would therefore be the preferred approach.

8.4.1 Comparative Effectiveness

8.4.1.1 Conformance with Project Treatment Standards (40 CFR 192)

Evaporation will result in complete removal of all contaminants from the aquifer, enabling total conformance with the project standards. Distillation will result in nearly total removal of hazardous constituents from the aquifer, and the treated water produced is expected to have a composition that will exceed the project standards and goals by one to two orders of magnitude with almost total removal of sulfate, nitrate, metals, and radionuclides. ZVI will remove arsenic, vanadium, and uranium essentially completely, but the treated effluent from the ZVI process may exceed the MCLs for selenium and molybdenum. This will limit the potential uses of this effluent. But the goal of the remediation is to enhance the effectiveness of the natural flushing process, and the ZVI process will meet this goal since it will remove vanadium to well below MCLs; the other constituents are expected to flush naturally within the statutory time limit.

All of the treatment processes can be designed to provide optimal protection of health for the plant operators and persons living or working in the vicinity, as well as those who depend on the alluvial aquifer for part or all of their water supply.

8.4.1.2 Ease of Residual Disposal

The only treatment residuals produced by the ZVI process are the iron itself, which will be disposed of at the end of the process; the effluent from the ammonia stripper, which will be purged as needed during the operation of the unit; and the tanks and miscellaneous process equipment, which should be available for reuse at the end of the project. The estimated volume of ZVI is 96 cubic ft, which will have a weight of about 12 tons assuming a porosity of 0.5. The amount of chemical required for the pH adjustment for the ammonia stripper will have to be

determined by laboratory testing, but it is expected to be less than 1 ton over the lifetime of the project.

The principal treatment residual produced by the evaporation and distillation processes is the concentrated sludge that contains the dissolved and suspended solids, which were removed from the ground water during treatment. The ground water contains the equivalent of 2,259 tons of sludge, based on the average TDS concentration. The evaporation process will produce about this amount of sludge over the lifetime of the project. The distillation process requires the addition of sulfuric acid, which will increase the sulfate concentration in the brine, as well as a small amount of antiscalant. This will increase the total sludge produced by the distillation process to about 2,325 tons.

The other major treatment residual from these two processes is the pond liners, which will be disposed of at the end of the remediation program. The spray evaporation process will produce four to five times as much of this material as the distillation process, because its pond is over four times larger.

Used piping, process equipment, filter elements, etc. which are discarded during treatment or are left over from the treatment systems at the end of the remediation, should be able to be free-released and disposed of at any commercial landfill operation, or reused elsewhere if the need exists. For this reason, estimates of the volume of such materials have not been made.

8.4.2 Comparative Implementability

8.4.2.1 Constructability

The simplicity of the ZVI system makes it the easiest of the three treatment processes to install. Both the spray evaporation and distillation processes require construction of a spray evaporation pond, which will require considerable skilled labor to install. The larger size of the pond required for the spray evaporation system will add cost but is not expected to add significantly to the difficulty of installation. The distillation treatment system is a self-contained unit and will be relatively simple to construct, requiring mainly field erection of the modules and some piping fabrication and connection. The distillation system vendor will handle all of this work.

8.4.2.2 Ease of Operation and Maintenance

As described above, this analysis assumes that a full-time operator will not be required for the ZVI system. A part-time GJO person visiting the site 3 to 4 times per week can handle operator duties for the ZVI system.

The distillation system is expected to be relatively easy to operate, because it is a packaged system designed to require minimal operator interface beyond routine monitoring. This treatment system will shut off automatically in the event of problems, and will relay the required information to the system monitor. The cost estimate for the distillation system assumes a single day-shift operator for operations and maintenance of the distillation system and the evaporation pond.

Operation of the spray evaporation system will not require a high degree of skill. However, the system is mechanically complex and covers a greater area, with more moving parts, than either

of the other treatment alternatives. It is therefore expected to require the most operator attention of any of the three treatment processes.

8.4.2.3 Expected Reliability

A less complex system is generally more reliable than a complex design. On this basis, the ZVI system would be expected to be the most reliable of the three treatment alternatives. The distillation system will be highly automated and will itself be very reliable, with less than 10 percent down-time expected for routine maintenance. However, the distillation system incorporates the spray evaporation system, which will be less reliable. The spray evaporation system in the evaporation process is expected to be less reliable than the evaporation pond used with the distillation system because of its larger size.

8.4.2.4 Ability to Handle Changes in Influent Composition

The spray evaporation system will remove all impurities regardless of concentration, since nothing is returned to the aquifer. If it becomes necessary to meet treatment standards for additional constituents, the effectiveness of either of the other processes will depend on the nature of the contaminant to be treated. Changes in influent concentration will affect the rate of brine generation in the distillation system, but since the contaminants in the ground water are not volatile, this system is expected to be reasonably tolerant of changes in influent. ZVI is judged least satisfactory from this standpoint; it will be effective against those impurities which can be removed by the reduction process, but will be ineffective against other contaminants.

8.4.2.5 Ability to Handle Increases in Extraction Capacity

The ZVI process is judged most satisfactory from this standpoint. The removal efficiency of ZVI appears to be a fairly weak function of residence time for most contaminants, so an increase in extraction rate should not greatly affect the performance of the ZVI reactor. The spray evaporation pond will be designed for a capacity of 120 gpm, and cannot easily be expanded. Increases in extraction rate may result in overfilling of the pond. If the pond does not overfill, it will have to operated for a period of time after extraction is complete, to achieve complete dewatering. The distillation unit will also be specified for a capacity of 120 gpm. Increasing the capacity beyond 120 gpm will require purchase and installation of additional unit(s).

8.4.3 Comparative Cost

The estimated capital cost, annual O&M cost, and present worth value for each of the individual treatment processes, as well as for the extraction process, have been given in their respective articles in Section 8.3, and are summarized below in Table 8–7. All costs are in millions of dollars.

Since the extraction system is identical for all three treatment processes, the costs for the three complete treatment alternatives can be calculated by adding the costs of the extraction system to each treatment process. This obviously does not affect the comparative ranking. And, as has been noted, the cost of the vanadium mobilization process is unknown and cannot be even roughly estimated. However, since it also is common to all treatment processes, it also does not affect the comparative cost rankings.

Table 8–7. Comparative Costs

Process	Capital	O&M (annual)	Present Value
Extraction with Vertical Wells	0.21	0.006	0.23
Spray Evaporation	1.20	0.37	2.18
Distillation	3.63	0.50	4.55
Zero-Valent Iron	0.076	0.057	0.18

8.4.4 Comparative Summary

The preceding discussion has presented nine evaluation criteria, and has compared the three pump-and-treat alternatives with regard to each of these criteria. Table 8–8 lists the treatment alternatives, in order of preference with 1 being the highest, for each of these ten evaluation criteria.

Table 8–8. Ranking of Pump-and-Treat Alternatives

Criterion	Treatment Alternatives in Descending Order		
	1	2	3
Conformance with Project Treatment Standards (40 CFR 192)	Evap	Dist	ZVI
Residual Disposal	ZVI	Dist	Evap
Constructability	ZVI	Dist	Evap
Ease of Operation and Maintenance	ZVI	Dist	Evap
Expected Reliability	ZVI	Dist	Evap
Ability to Handle Changes in Influent Composition	Evap	Dist	ZVI
Ability to Handle Increases in Extraction Capacity	ZVI	Evap	Dist
Comparative Cost—Initial Capital Outlay	ZVI	Evap	Dist
Comparative Cost—Present Worth	ZVI	Evap	Dist

Key:
Evap = Spray Evaporation
Dist = Distillation
ZVI = Zero-Valent Iron

In-situ stabilization of vanadium is not included in this table because its technical feasibility has yet to be established. If it were demonstrated to be feasible, it would be expected to be less costly than any of the pump-and-treat processes both in initial capital and in present worth (since it has no operating cost), and would be at least as good as the best of the pump-and-treat processes from all the other criteria as well.

8.4.4.1 Determination of Proposed Treatment Process

Treatment Alternative 3—Zero-Valent Iron with Ammonia Stripping, is the winner for seven of the nine categories. It offers the lowest initial capital cost, as well as the lowest total project cost, by a significant margin. It also is the preferred technology from the standpoints of residual disposal, ease of operation and maintenance, expected reliability, and ability to handle changes in extraction capacity. On the other hand, it is the least desirable treatment alternative for conformance with treatment standards and goals, and ability to handle changes in influent composition. These are significant problems for this technology, because the fact that ZVI cannot meet treatment standards for at least one constituent (molybdenum) and may have trouble with another (nitrate) will limit the possible uses for the treated water.

Treatment Alternative 1—Spray Evaporation, is the winner for the two criteria that ZVI did not win—conformance with treatment standards and goals, and ability to handle increases in influent composition. It finished last in four categories—residual disposal, constructability, ease of operation and maintenance, and expected reliability. None of these is particularly serious, given the short-term nature of the project.

Treatment Alternative 2—Distillation, is not the first choice for any of the evaluation criteria. It is by far the most expensive, both to install and to operate, though much of the capital cost could be recovered if the distillation unit could be utilized at another UMTRA ground water site after the remediation is complete at New Rifle. This is by no means a remote possibility, since the distillation system proposed is similar, both in type and in capacity, to the units that will be installed at both the Tuba City and Monument Valley sites. Even if this could be done, however, distillation would still be the most expensive alternative, though it might not be significantly more costly than spray evaporation. Distillation was also the least desirable option for ability to handle increases in extraction capacity. This may not be a serious objection; if treatment of more than three pore volumes is required to meet the treatment standards, remediation at New Rifle may be judged technically nonfeasible.

Of the three processes, distillation is the poorest choice, combining high cost with mediocre performance on the other evaluation criteria. Spray evaporation is a better performer, and would be the option of choice if there were no readily available option for disposing of the treated water from the ZVI process. The inability of the ZVI technology to meet treatment standards for at least one constituent complicates the disposal situation. Of the five non-evaporative disposal options presented in Section 8.2.2 (discharge to wetlands, discharge to surface water, injection wells, discharge to municipal water treatment facility, and sale to municipal water supply) the effluent from the ZVI process might be suitable for three (discharge to wetlands, injection wells, or discharge to municipal water treatment facility). However, additional laboratory work would be required to improve the characterization of the process and fine-tune the likely effluent composition, and further discussions with regulators and other stakeholders would have to take place to confirm the acceptability of the treated water for the disposal option selected.

Nonetheless, the low cost and technically innovative nature of the ZVI process argue for its being the treatment option of choice. It is the winner in seven of the nine evaluation criteria listed, and the chances are excellent that its effluent will be suitable for at least one of the three possible disposal options listed above. It makes virtually no waste and requires very little in the way of operating manpower. In consideration of all the above, then, and with the caveat that the disposal of the treated water must be resolved before a final selection can be made, Treatment Alternative 3—Zero-Valent Iron with Ammonia Stripping, tentatively must be considered the preferred treatment technology for the New Rifle ground water remediation program.

However, even the ZVI process is probably more expensive than in-situ stabilization of the vanadium. This process will be investigated during a series of laboratory and field studies that are planned for fiscal year 2000. These studies will also investigate the feasibility of vanadium mobilization, and will obtain additional data on the ZVI process and the ammonia stripping process. Should the in-situ stabilization process prove technically feasible, discussions with regulators and stakeholders will be required to determine whether implementation of this process is “politically” acceptable.

8.5 Proposed Remediation Process

The proposed remediation process consists of three systems. This section discusses each of those systems.

8.5.1 Description of Proposed Remediation Process

8.5.1.1 Proposed Extraction System

The extraction system consists of a total of nine extraction wells, varying in depth up to a maximum of approximately 20 ft, depending on the depth of the alluvial aquifer at the particular location. The expected flow rate per well is 15 gpm, giving the extraction system a peak capacity of 135 gpm once all wells are in service. The design provides one extra well for increased reliability.

A typical extraction-well design for the New Rifle site would consist of a 10-inch diameter borehole completed with 6-inch diameter wire-wrapped well screen and blank PVC riser. The section of the well containing the well screen will be completed with an appropriately-sized sand pack. The final design of the well and the size of the pump will be optimized based on field conditions.

The extraction wells will be installed across the vanadium plume, an area measuring approximately 1 million square ft or about 23 acres. The water pumped from these wells must be collected from across this substantial area and delivered to the treatment facility. Each pump will discharge into a 4-inch PVC outlet pipe. These outlet pipes will be directed into a series of headers which, in turn, will connect to the main 4-inch PVC extraction system discharge pipe, which is routed to the treatment facility.

The vanadium mobilization system will also be installed across the vanadium plume. No design details of this system can be provided at this time.

8.5.1.2 Proposed Treatment System

The water from the extraction system will be fed to the ammonia stripper. Sodium hydroxide or another base will be added to raise the pH and increase ammonia volatility. The stripper will be either an agitated tank or a tank with a circulating column. Laboratory work will be required to optimize residence time and chemical addition.

From the ammonia stripper, contaminated water is pumped directly to the ZVI reactor, which will consist of a single tank with a capacity of about 1,000 gallons. The tank will be filled with about 96 cubic ft of ZVI, which will give the system a residence time of about 6 minutes at the design influent rate of 120 gpm.

8.5.1.3 Proposed Disposal System

Water from the ZVI reactor will be pumped continuously to disposal. The nature of the disposal system will be determined after completion of the additional laboratory work on the ZVI process, as discussed above.

8.5.2 Summary

The proposed system meets or exceeds the requirements of 40 CFR 192, and is protective of human health and the environment. The only product of the treatment system is the treated water produced by the ZVI reaction. The only residuals are the ZVI and small quantities of chemicals, which will be purged as needed (chemicals) or disposed of at the conclusion of the remediation project (ZVI).

8.5.3 Limitations of Proposed Alternative

The proposed treatment process depends on several technologies which have not been demonstrated or that have not been adequately characterized. The extraction process depends on development and field implementation of an effective technology for mobilizing vanadium. The ZVI process requires additional characterization to optimize its effectiveness for treatment of as many of the contaminants as possible. Finally, as has been stated elsewhere, if the technical feasibility of in-situ stabilization can be demonstrated, it would become the preferred process. Treatment cannot begin until these technological issues have been resolved.

Although ground water extraction and ex-situ treatment, also known as pump and treat, was found to be the best method to meet cleanup goals in the aquifer, the effectiveness of pump-and-treat systems has been limited. Few sites with contaminated ground water have ever been restored to drinking water standards (Travis 1990; EPA 1996); however, the vast majority of sites where pump and treat is now being used are dealing with sources composed of non-aqueous-phase liquids.

The effectiveness of the ground water extraction system, and in particular the vanadium mobilization process, is the primary factor that will determine whether aquifer cleanup goals can be met. Laboratory and pilot studies will be performed at the site to attempt to demonstrate the mobilization process for vanadium. 40 CFR 192.21(f) allows the application of supplemental standards in the event that restoration of ground water is technically impracticable from an engineering perspective. This would only be considered for the New Rifle site if the laboratory and pilot studies are unable to demonstrate increases in vanadium mobilization that would be sufficient to substantially reduce vanadium levels in the aquifer to the point where the remediation can be completed by natural flushing.

End of current text

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