

WELLFIELD RESTORATION REPORT CHRISTENSEN RANCH PROJECT WYOMING

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1 INTRODUCTION

COGEMA Mining, Inc. (COGEMA) has completed the aquifer restoration program at the five Mine Units (MUs) where mining has been completed within the Christensen Ranch in-situ-leach (ISL) permit area. The Christensen Ranch ISR mine is permitted under the Wyoming Department of Environmental Quality (WDEQ) Permit to Mine No. 478 and the U.S. Nuclear Regulatory Commission (NRC) source material license No. SUA-1341. This report provides the aquifer restoration history and supporting data to demonstrate the successful restoration of these MUs.

The report is structured to first provide an overall introduction to the Christensen Ranch project and the restoration process, then individual restoration reports for each mine unit are provided. Consistency in the geology and hydrogeology across the Christensen Ranch permit area justifies submittal of a single report for purposes of demonstrating completion of restoration. In the future, after other areas are mined at Christensen, the report can be augmented with additional restoration reports for those mine units.

Wellfield restoration operations were initiated at Mine Units 2, 3 and 4 (MU2, MU3, and M4, respectively) in 1997 and in Mine Units 5 and 6 (MU5 and MU6) in 2000. Restoration of all MUs, including stability monitoring, was completed by 2006. An average of 10.1 pore volumes of water were treated for the five Christensen Ranch MUs during restoration activities. Groundwater within the production zone has been restored to the pre-mining class of use, using Best Practicable Technology (BPT), as required by the WDEQ.

Baseline (pre-mining) water quality was used to establish Target Restoration Values (TRVs) for thirty-five parameters in each MU. TRVs are compared against the mean wellfield concentration measured within a specific MU during stability monitoring. Restoration operations have resulted in water quality that meets the TRVs for most of the parameters. Of the parameters that did not achieve the TRV, most do not have WDEQ or US Environmental Protection Agency (EPA) regulatory standards. In some cases, a parameter exceeded the TRV, but remained below WDEQ or EPA standards.

The only constituents that exceeded TRVs and either WDEQ or EPA standards in at least one MU were iron (Fe), manganese (Mn), selenium (Se), total dissolved solids (TDS), uranium (U) and radium-226 (Ra-226). Tables 1.1 and 1.2 summarize results of the wellfield restoration relative to the TRVs.

Stability monitoring, completed in 2006, demonstrates no strongly increasing concentration trends for any of the 35 monitored constituents, indicating that post-restoration conditions are stable.

This report, including supporting data and discussion demonstrates that groundwater restoration at the Christensen Ranch in-situ uranium mine meets the requirements for unconditional restoration approval by WDEQ and NRC.

Table 1.1 Summary of Restoration Results by Mine Unit, Christensen Ranch

Category	MU2	MU3	MU4	MU5	MU6
Total Number of Monitored Parameters	35	35	35	35	35
Number of Parameters < TRV	24	27	21	25	27
Number of Parameters > TRV but with no WDEQ Class I or EPA MCL standards	5	5	6	4	4
Number of Parameters > TRV but < WDEQ Class I or EPA MCL standards	2	2	2	3	1
Number of Parameters > TRV and > WDEQ Class I or EPA MCL standards	4	1	6	3	3

TRV – Target Restoration Value

WDEQ Class I-Drinking water use

EPA MCL – Maximum Contaminant Limit

Table 1.2 Parameters Exceeding TRV and WDEQ or EPA Standards by Mine Unit

Mine Unit	TDS	Fe	Mn	Se	U	Ra
MU2		X	X		X	X
MU3			X			
MU4	X	X	X	X	X	X
MU5			X	X	X	
MU6			X	X	X	

X- Parameter exceeds TRV and WDEQ or EPA standards in specified MU

1.1 BACKGROUND

COGEMA operates the Christensen Ranch in-situ leach (ISL) uranium mine in Johnson and Campbell Counties, Wyoming (Figure 1-1). Operation of the mine is incorporated under Wyoming Permit to Mine 478 and NRC Source Material License SUA 1341. The Christensen Ranch operation includes surface facilities and a satellite treatment plant, well houses, trunklines/pipelines, and mining and monitoring wells typical of ISL operations.

1.2 PURPOSE AND OBJECTIVES

The purposes of this report are:

- to discuss the significant beneficial results from restoration operations performed at Christensen Ranch;
- to document that COGEMA's restoration activities at Christensen Ranch have resulted in groundwater quality consistent with the pre-mining class of use; and,
- to provide the supporting discussion and data such that WDEQ and NRC have a strong basis for granting unconditional approval of the Christensen Ranch restoration activities.

1.3 REPORT ORGANIZATION

This report is structured to initially present generalized information regarding the Christensen Ranch ISL project, including: regulatory history (Section 2), site description (Section 3), geology (Section 4), hydrology (Section 5), general water quality (Section 6), and a summary of restoration objectives and methods (Section 7). More detailed information for the individual MUs is included as Restoration Data Packages (RDPs) for each MU, which are included as attachments to the main body of this report. The RDPs include:

- a description of localized geology and hydrogeology;
- identification of MU-specific baseline water quality, target restoration values and groundwater classifications;
- description of permitting, mining and restoration activities specific to the MU; and,
- a summary of the MU restoration results.

RDPs are included for MU2, MU3, MU4, MU5 and MU6, which are the only mine units operated at the Christensen Ranch to date. Specific water quality data pertinent to each of the MUs are included as Appendices to the respective RDPs.

Section 8 of this report includes summary and conclusions regarding the effectiveness of the restoration activities for all of the MUs.

2 REGULATORY ISSUES

2.1 PERMITTING HISTORY

Malapai Resources Company permitted the Christensen Ranch ISL project as an amendment to Permit to Mine No. 478. Permit to Mine No. 478 was originally issued by the WDEQ Land Quality Division (LQD) on August 18, 1978 for the Irigaray Mine Site. Amendment A2 to incorporate Christensen Ranch was submitted by Malapai and approved by WDEQ in 1988. The NRC Source Material License SUA-1341, originally issued in November 1978 for the Irigaray mine, was amended in 1988 to incorporate Christensen Ranch. Commercial production at Christensen Ranch MU3 was approved by WDEQ in January 1989 and commenced in March 1989.

Numerous amendments have been issued to the NRC license since 1978. In 1996 COGEMA submitted to both WDEQ and NRC an updated permit document (NRC renewal application). This essentially combined the critical elements of the original Irigaray permit document with the Christensen Ranch permit documents. This document was subsequently accepted by both agencies in 1997.

2.2 REGULATORY FRAMEWORK

2.2.1 Groundwater Classification

In November 2001 a joint WDEQ-LQD and Water Quality Division Advisory Board policy was issued with regard to in-situ groundwater classification and restoration. In this policy it was recognized that treating a groundwater source that contains radium at background concentrations commonly found in a uranium production zone could produce a solids filtrate or wastewater that would be prohibited for unrestricted release. Therefore, the concept of treatability for radium-226 no longer appeared applicable. The policy, dated November 14, 2001, specifically states that radium will not be considered as treatable due to concerns with the safe disposal of any water treatment by-products (applies to Section 5 of WDEQ-WQD Chapter VIII rules and regulations). Because the radium-226 concentrations at baseline within the Christensen Ranch MUs were all above 5 pCi/l (Class I, II and III standards), the applicable groundwater classification for all these units is Class IV.

2.2.2 Restoration Law and Regulations

Groundwater restoration is defined in the Wyoming Statutes [W.S. §35-11-103(f)(iii)] as "the condition achieved when the quality of all groundwater affected by the injection of recovery fluids is returned to a quality of use equal to or better than, and consistent with the uses for which the water was suitable prior to the operation by employing the best practicable technology." Best Practicable Technology (BPT) is further defined in W.S. §35-11-103(f)(i) as "a technology-

based process justifiable in terms of existing performance and achievability in relation to health and safety which minimizes, to the extent safe and practicable, disturbances and adverse impacts of the operation on human or animal life, fish, wildlife, plant life and related environmental values." The WDEQ-LQD rules and regulations regarding ISL (Chapter XI) also state that the standard for restoration is to return the groundwater to the pre-mining class of use.

2.3 GROUNDWATER CLASSIFICATIONS, AQUIFER EXEMPTION

The basis for the UIC program in Wyoming is WDEQ's groundwater classification system provided in Chapter VIII of Water Quality Division's rules and regulations. Recent changes to these regulations codify the November 2001 policy that radium-226 will not be considered treatable. Therefore, all mine unit water quality is considered Class IV pursuant to the WDEQ regulations. However, prior to the 2001 policy and recent regulation changes, the WDEQ Water Quality Division did an initial groundwater classification based on the old regulations. These are provided below, but it must be emphasized that these classifications are outdated and each Mine Unit's true classification is Class IV.

◆ MU2

- All groundwater in the ore zone was determined by WDEQ 1993 to be Class I, primarily based on sulfate and TDS concentrations. However, all of the 25 baseline ore zone wells exceeded the radium standard for Class I, II and III classification. Based on the revised WDEQ regulations that radium will not be considered as treatable due to concerns with the safe disposal of any water treatment by-products, the applicable groundwater classification for the MU2 ore zone wells is Class IV.

◆ MU3

- Groundwater classifications were determined for this MU by WDEQ in 1989. Nineteen wells were used for the classification. All groundwater in the ore zone was determined by WDEQ-LDQ to be Class IV. However, WDEQ-WQD had mixed classifications on a well-by-well basis. As with MU2, all ore zone wells exceeded the Class I, II and III standard for radium and should be classified as Class IV.

◆ MU4

- In 1994, WDEQ classified all 12 wells used for ore zone baseline determination as Class I. Each of the wells had radium-226 above the Class I, II and III classification standard of 5 pCi/L thereby indicating Class IV as the appropriate classification.

◆ MU5

- In 1995, WDEQ provided a mixed classification of the 25 ore zone baseline wells. Twelve of the wells were classified as Class IV and thirteen of the wells as Class I. The Class IV designation was primarily based on TDS and radium concentrations (for radium in excess of 100

pCi/L). The data from the baseline monitoring shows that all wells exceeded 5.0 pCi/l and should be Class IV groundwater.

◆ **MU6**

- All of the ore zone monitor wells except two were classified by WDEQ as Class IV in 1996. The remaining two wells were classified as Class III. However, using the current WDEQ policy for radium, those two wells should also be designated as Class IV groundwater.

In summary, pre-mining water quality samples from all baseline ore zone monitor wells at Christensen Ranch (MU2 through MU6) contained radium-226 that exceeds Class I, II, and III standards. The appropriate classification for those wells is Class IV for industrial use.

3 SITE DESCRIPTION

This section of the report summarizes the general site conditions at Christensen Ranch including location, physiography, topography and climate. Additional information regarding site conditions can be found in the Christensen Ranch Amendment Application (Malapai 1988). Details concerning specific MUs are described in the attached RDPs.

3.1 LOCATION

The Christensen Ranch Permit Area covers approximately 14,000 acres in Johnson and Campbell Counties, Wyoming. The Permit Area is located in Townships 44 and 45 North, Ranges 76 and 77 West, approximately 95 miles northeast of Casper, 50 miles southeast of Buffalo, and 45 miles southwest of Gillette, Wyoming (Figure 3-1), in the west central part of the Powder River Basin, Wyoming. The facilities are located 13 road miles southeast of the Irigaray Mine. MUs 2 through 6 are located in Sections 3, 4, 6, 7, 9, 10, 16, 17, 18, 19, 20 and 21, of T44N, R76W (Figures 3-2). The total area within the five MUs is approximately 200 acres.

3.2 PHYSIOGRAPHY AND TOPOGRAPHY

The Christensen Ranch site is located in the west-central portion of the Powder River Basin (PRB). The site is within the Great Plains Physiographic Province, which is characterized by broad river plains and low plateaus on stratified sedimentary rocks.

The topography of the central PRB is dominated by plains, rolling hills, and tablelands. Topographic relief has resulted from structural deformation on the west, east and south edges of the Basin and historical deposition and erosional cycles within the Basin itself. On a regional basis, the surface of the Basin sediments dips gently (1 to 2 degrees) to the north-northwest.

Locally, the elevation within the Christensen Ranch MUs ranges from approximately 4,550 to 4,950 feet above mean sea level (ft amsl). The area is characterized by gently rolling hills with deeply dissected drainages. The land surface elevation east of the MUs increases over a three to four mile distance to approximately 6,050 ft amsl at North Butte (Figure 3-2). To the west-northwest, the land surface slopes downward over an eight-mile distance to the Powder River at approximately 4,220 ft amsl.

4 GEOLOGY

Regional and site geology are briefly summarized in this section of the report. A more detailed discussion of geologic conditions for the Christensen Ranch project can be found in the Christensen Ranch Amendment Application (Malapai, 1988). Additional details pertinent to the site-specific geologic conditions at individual MUs are found in the attached RDPs.

4.1 REGIONAL GEOLOGY

The Powder River Basin is a large, asymmetrical structure bounded on the west by the Big Horn Mountains, on the east by the Black Hills, and by the Laramie Range and Hartville Uplift on the south. The basin orientation and plunge generally are south to north. The basin is open to the north and encloses approximately 13,000 square miles of northeastern Wyoming (Sharp & Gibbons, 1964). The Christensen Ranch Mine is located east of the axis of the PRB.

The Precambrian basement within the PRB is overlain by approximately 16,000 to 18,000 feet of sediments. Lower to Upper Cretaceous-age rocks (Frontier to Lance Formations) outcrop on the west side of the PRB, whereas predominately Upper Cretaceous (Lance) and Pliocene (Fort Union) deposits are present on the south and east sides (Hodson, et. al., 1973). In the central portion of the basin, these marine and fluvial deposits are approximately 8,000 feet below the outcrop elevations. Eocene Wasatch Formation is present at the surface across most of the PRB. Dips of beds within the Wasatch section range from less than one degree to 2 ½ degrees.

4.2 SITE STRATIGRAPHY

The specific stratigraphy that applies to this study includes the section from the Oligocene White River Formation to the Paleocene Fort Union Formation. A generalized stratigraphic section for the project area is shown in Figure 4.1. These deposits are described, in descending sequence, in the following sections.

4.2.1 Surficial Deposits

Surficial deposits in the vicinity of Christensen Ranch consist mainly of weathered sediments of the Wasatch Formation, alluvium in small drainages, and alluvium/terrace deposits in the immediate vicinity of the Powder River. The general character of the alluvium has been described by Love and Christiansen (1985) as clay, sand, silt, and gravel present in flood plains, fans, terraces, and slopes. A more detailed presentation of surface geology specific to the vicinity of the Powder River drainage is provided by Ringen and Daddow (1990).

The Oligocene White River Formation, which is commonly found on the surface in the fringes of the PRB, has been eroded away in the Christensen Ranch area.

Occasional surficial deposits are encountered in the vicinity of Pumpkin Buttes. The White River is composed of tuffaceous sandstone, conglomerate, and claystone.

4.2.2 Wasatch Formation

The Eocene-age Wasatch Formation unconformably overlies the Fort Union Formation. It is present on the surface throughout the Christensen Ranch area, and most of the central portion of the PRB. The Wasatch is comprised of claystone, lenticular sandstone, and minor coal deposits of fluvial origin. In the vicinity of Christensen Ranch, the Wasatch varies from 650 to 1,000 feet thick.

Within the Christensen Ranch area, the Wasatch formation has been divided into three units. The units, referred to as the "J", "K", and "L" fluvial systems in descending order, are briefly described below. Detailed descriptions of each unit, including structure and isopach maps are provided in the Christensen Ranch Amendment Application (Malapai, 1988). Table 4.1 summarizes the relationship between the units of the Wasatch Formation at Christensen Ranch.

The "J" Fluvial System is defined as the stratigraphic interval from the base of the upper aquitard that overlies the "K" fluvial system to the ground surface within the Christensen Ranch area. The interval is dominated by siltstones and mudstones with thin sandstone units deposited. The total thickness of the "J" Fluvial System is variable, ranging from 300 to 450 feet. The base of the "J" Fluvial System includes a 90- to 100-foot thick shale/claystone interval that overlies the uppermost uranium bearing sandstone within the "K" fluvial system. This facies acts as the confining unit between the "K" and "J" sandstone aquifers and is defined as the Upper Aquitard.

The "K" Fluvial System is composed of sandstones that are the primary hosts for the uranium ore bodies of the Christensen Ranch project. The "K" Fluvial System is defined as the stratigraphic interval from the top of the Lower Aquitard that overlies the "L" Fluvial system to the base of the Upper Aquitard that marks the beginning of the "J" Fluvial System. The thickness of the "K" Fluvial System is typically 150 to 200 feet within the Christensen Ranch area and includes sandstones, shales and mudstones. Porosity estimates provided in the indicated 26% for the ore-bearing portion of the aquifer and 29% for the non-ore bearing part of the aquifer.

The lowermost unit is the "L" fluvial system. It is defined as the stratigraphic interval between the Fort Union/Wasatch contact and the base of the lowest uranium mineralized host sandstone (Taucher 1987). The "L" fluvial system consists of one to two continuous sandstone aquifers separated from each other and from the overlying and underlying units by shales, mudstones and siltstones. The "L" fluvial system is approximately 200-230 feet thick within the developed Mine Units. The uppermost portion of the "L" fluvial system is a 60- to 70-foot

thick shale/mudstone interval that is the confining layer separating aquifers of the "L" and "K" fluvial systems. This unit is defined as the Lower Aquitard and is continuous through the Christensen Ranch area.

A generalized stratigraphic column through the Wasatch Formation is provided below.

Table 4.1. Localized Stratigraphic Column and Description Surface through the "L" Fluvial System, Christensen Ranch Mine		
<i>Geologic Section (Classification)</i>	<i>Typical Thickness (Feet)</i>	<i>Description</i>
"J" Sandstone	200-350	Interbedded, discontinuous sandstone, siltstone and shales
Upper Aquitard (Base of the "J" Fluvial System)	90-100	Interbedded gray siltstones and claystones with a laterally extensive lignite seam
"K" Sandstone	150-200	Massive sandstone to interbedded sandstones, siltstones and shales
Lower Aquitard (Top of the "L" Fluvial System)	60-70	Clayey siltstones and shales
"L" Sandstone	140-160	Discontinuous sandstone stringers interlayered with siltstones and shale

5 HYDROGEOLOGY

Regional and site hydrogeology are briefly summarized in this section of the report. A more detailed discussion of hydrogeologic conditions for the Christensen Ranch project can be found in the Christensen Ranch Amendment Application (Malapai 1988). Additional details pertinent to the hydrogeologic description of individual MUs are found in the attached RDPs.

5.1 REGIONAL HYDROGEOLOGY

Historical studies have stated that regional flow systems (e.g., the Wasatch, Fort Union, and deeper sediments) generally flow to the northern portion of the PRB and discharge via unknown avenues in Montana (Lowry & Wilson, 1986, and Rankl & Lowry, 1990). The hydraulic communication between the flow systems has been reported to vary from none to direct. Flow direction in those sediments near outcrop areas generally has been characterized as toward the center of the PRB.

On a semi-regional scale, groundwater flow occurs to the north-northwest, and the gradient is on the order of 0.004 to 0.006 ft/ft. This ground-water flow direction is consistent with results numerous studies (Honea, 1974; Morris & Bahr, 1975; NRC, 1978; Rose, 1971) on a scale specific to Christensen Ranch.

5.2 SITE HYDROGEOLOGY

As discussed previously in Section 4, the hydrostratigraphic units of primary interest in analyzing the results of restoration operations at Christensen Ranch are within the Wasatch Formation and include the following:

- The "J" Sandstone is the first aquifer above the "K" Sandstone and the Upper Aquitard. Groundwater within this unit is under confining conditions.
- The Upper Aquitard separating the "J" Sandstone and the "K" Sandstone.
- The "K" Sandstone is the uranium producing interval at Christensen Ranch. Groundwater within this unit is under confining conditions.
- The Lower Aquitard separates the "K" Sandstone and the "L" Sandstone
- The "L" Sandstone is the first water bearing zone beneath the "K" Sandstone. Groundwater within the "L" Sandstone is under confining conditions.

5.2.1 Potentiometric Surface, Flow Direction and Hydraulic Gradient

Water-level data collected from site monitor wells prior to mining activities indicate that the natural, unstressed groundwater flow direction within the "J", "K" and "L" Sandstones at the Christensen Ranch site is generally to the northwest. However, horizontal hydraulic gradients vary between the "J", "K" and "L" Sandstones and differences in water levels demonstrate the lack of hydraulic communication between them.

Potentiometric surface maps for the "J", "K" and "L" Sandstones, generated from water level data collected in the fall of 1986, were included in the Christensen Ranch Amendment Application (Malapai Resources, 1988)

The hydraulic gradient in the "J" Sandstone is generally to the west along the eastern portion of the permit area (at 0.01 ft/ft) but becomes more northwesterly (at 0.008 ft/ft) toward the northwest edge of the Christensen Ranch permit boundary.

The hydraulic gradient in the "K" Sandstone throughout the wellfield area is to the northwest at 0.005 to 0.009 ft/ft. Water levels are approximately 30 feet lower within the "K" Sandstone than in the "J" Sandstone in the area of mine units MU5 and MU6.

The hydraulic gradient for the "L" Sandstone is also to the northwest, at approximately 0.005 to 0.007 ft/ft. Water levels in the "L" Sandstone are generally 10 to 30 feet lower than in the "K" Sandstone, indicating hydraulic separation between the units.

5.2.2 Aquifer Properties

Based on the site stratigraphic sections, water levels in the "J", "K" and "L" Sandstones, and pumping tests conducted during the mine permitting process, the "K" Sandstone has been shown to be a confined aquifer in the vicinity of Christensen Ranch.

Nine aquifer pumping tests were performed at six hydrologic test sites within Christensen Ranch in the K Sandstone between 1977 and 1986 as part of the mine permitting process. Details of those tests are described in the Amendment Application (Malapai 1988).

Based on review of the testing data, the geometric mean transmissivity (T) for the "K" Sandstone ranged from 264 to 1030 gpd/ft (33 to 138 ft²/d); the hydraulic conductivity (k) varied from 0.32 to 0.54 ft/d. and the storativity (S) ranged from 8.7 E-05 to 1.5 E-03. Aquifer parameters determined from specific mine unit testing are described in the Mine Unit Attachments.

Because of the lack of response within the "J" and "L" Sandstones during the pumping tests, no aquifer parameters were determined for those hydraulic units. However, because the lithology of the "J", "K" and "L" Sandstones are similar, and the hydraulic gradients in each of these units are of similar direction and magnitude, it may be surmised that aquifer properties in the "J" and "L" Sandstones are of similar magnitude to those of the "K" Sandstone.

The horizontal permeability of the upper and lower confining units reported by D'Appolonia (1983) and Canonie (1986), are less than 10^{-6} cm/sec (approximately 0.0028 feet/day) and vertical permeability was calculated as less than 10^{-8} cm/sec.

Additional aquifer testing was performed at each of the MUs prior to commencement of production. Results of the aquifer tests are discussed in the Wellfield Data Packages that were submitted for each MU (Malapai, 1988, Total Minerals Corporation 1991 1992, 1993a and 1993b, COGEMA 1994a, 1994b, 1995 and 1996) and are summarized in the MU RDPs attached to this report.

6 WATER QUALITY

This section of the report describes the general water types that occur within the Christensen Ranch Permit Area in the zones of interest, and how baseline water quality/classification for the individual MUs is determined.

6.1 GENERAL WATER TYPE

A regional groundwater quality sampling program was conducted for the Christensen Ranch Project as part of the mine permitting process to determine general baseline water quality as described in the Christensen Ranch Amendment Application (Malapai, 1988). Samples were collected from the production zone ("K" Sandstone), first overlying aquifer ("J" Sandstone), and first underlying aquifer ("L" Sandstone) at ten locations spaced across the Permit Area. Figure 6.1 shows the locations of the regional groundwater quality monitor wells. Four sampling episodes were conducted from 1986 through 1987. Results of the sampling indicated that the "K" Sandstone water type is predominately sodium sulfate whereas the "J" Sandstone ranged from sodium bicarbonate to sodium calcium-sulfate and the "L" Sandstone ranged from sodium bicarbonate to sodium sulfate water type. There is greater variability in general water type in the "J" and "L" Sandstones than in the K Sandstone. The variability is likely a function of the discontinuous nature of the "J" and "L" Sandstone, compared to the "K" Sandstone.

6.2 WATER CLASSIFICATION/BASELINE QUALITY

As reported in the Christensen Ranch Amendment Application (Malapai, 1988), water quality analysis for the regional groundwater-sampling program consistently exceeded either WDEQ or US Environmental Protection Agency (EPA) standards. Of the ten production zone ("K" Sandstone) wells included in the sampling program, eight did not meet the WDEQ Class I (Domestic Use) standard, none met the WDEQ Class II (Agricultural Use) standard, and only one met the Class III (Livestock Use) standard. Sulfate, TDS and pH were the primary parameters that exceeded the standards. Six of the production zone wells did not meet the EPA standard for drinking water, with radium-226 being the most common parameter causing exceedances. Similarly, water quality in all of the "J" and "L" Sandstone wells failed to meet standards for either Wyoming Class I, II or III groundwater.

In addition to the regional sampling program, numerous groundwater samples were collected from 24 "K" Sandstone wells, one "J" Sandstone and one "L" Sandstone well at the Willow Creek Research and Development (R&D) Site between 1982 and 1986. As with the regional program, WDEQ and EPA standards were frequently exceeded by the water quality results from the Willow Creek R&D wells.

Details of the regional and Willow Creek R&D water quality sampling programs are provided in the Christensen Ranch Amendment Application (Malapai, 1988).

The baseline water quality data point to the fact that water quality in the Christensen Ranch project site is of generally poor quality and seldom meets domestic classification standards prior to uranium mining activities.

6.3 MINE UNIT BASELINE QUALITY

Baseline water quality was determined for each MU prior to commencement of production. Baseline water quality was determined within the production zone of the MU, on the perimeter of the production zone, and in the overlying and underlying aquifers. The results of the baseline water quality data collection and assessment were submitted to WDEQ and NRC as Wellfield Data Packages (WDPs) for each MU prior to production. Results provided in the WDPs are summarized in the attached RDPs.

Baseline water quality within the production zone is used as the basis for determining groundwater restoration goals for individual MUs. Ore zone baseline water quality was established by sampling designated restoration wells four times, separated by a minimum of two weeks. The restoration well density was one well per acre of wellfield. Two Wyoming Guideline 8 analyses and two short list analyses were conducted on the samples. A total of 35 parameters were included in the analysis. Target Restoration Values (TRVs) were calculated from the ore zone baseline water quality, as described in the following section, and submitted to WDEQ and NRC for approval prior to commencing production.

Wellfield Data Packages (WDPs) were submitted prior to commencement of production for each of the MUs. The WDPs include the following information:

- baseline water quality data and proposed monitor well UCLs,
- locations and completion details for monitor wells and ore zone baseline water quality wells,
- average mine unit baseline water quality and proposed TRVs,
- a demonstration that perimeter monitor wells are in communication with the ore zone wells, and
- pre-mining potentiometric surface maps of the production zone and overlying and underlying aquifers.

7 RESTORATION OBJECTIVES, TRVS AND METHODS

This section of the report provides an overview of the restoration objectives and the methods used to achieve those objectives at the Christensen Ranch ISL project. Information specific to individual MUs are provided in the respective RDPs for each MU.

7.1 RESTORATION OBJECTIVES

The first objective of groundwater restoration in the Christensen Ranch Permit No. 478 and NRC License SUA 1341 following production of an ore body is to return groundwater quality to baseline. TRVs were established for each MU based on the results of groundwater baseline quality sampling. If baseline quality cannot be achieved using BPT, then groundwater must be restored to pre-mining class of use.

7.2 TARGET RESTORATION VALUES

TRVs for the monitored constituents were calculated for each MU based on the results of the baseline water quality sampling. The TRVs were calculated on a MU specific basis using the tolerance limit method described in Section 5.8 of the Permit to Mine 478, A-2 Update (COGEMA 1996). TRVs are a function of the wellfield average baseline, the range of baseline results, and the baseline variability after removal of outliers. TRVs fall between the tolerance limits calculated for the MU baseline. The tolerance limits provide a baseline range with an acceptable low, mean and high value. The acceptable range for the baseline calculated with the tolerance test is as follows:

$$\bar{x} \pm Ks$$

where: \bar{x} = sample mean
K = tolerance factor, corresponding to $\beta = 0.99$ and $\alpha = 0.001$
s = sample standard deviation

TRVs for each of the MUs are summarized in Table 7-1.

7.3 RESTORATION METHODS

Best practicable technology (BPT) was applied throughout the Christensen Ranch groundwater restoration program. The process employed was completely justifiable in terms of performance and achievability in relation to health, safety and minimization of adverse impacts to the environment.

The restoration program followed the approach provided in Section 6 of the Permit to Mine No. 478, A-2 Update (COGEMA 1996). The phases were:

1. Groundwater Sweep
2. Permeate Injection (with reductant addition as necessary)

3. Aquifer Recirculation
4. Post-Restoration Stabilization

In the first three 'process' phases, the groundwater was actively recovered and treated as necessary for re-injection or disposal. The final phase involved the monitoring of designated restoration wells to assess the restoration efforts and the stability of the ore-zone water quality.

The first phase of restoration, groundwater sweep, was conducted to recall the mining solution (lixiviant) from the affected aquifer (termed ore or mineralized zone) and from the area surrounding the ore zone that may have been affected by flaring of lixiviant during mining. Groundwater sweep involves the 100% consumptive removal of the wellfield groundwater with no re-injection. The process is termed groundwater sweep because the removal of water from the ore zone creates a hydrologic sink, wherein the surrounding groundwater "sweeps" into the mined ore zone, recharging the zone with native connate water. The water recovered during groundwater sweep was treated for discharge to either a NPDES outfall (Permit No. WY03642) to on-site evaporation ponds, or discharged to two Class I injection wells (WDEQ Permit Number UIC 00-340).

The second restoration phase, Reverse Osmosis (RO) permeate injection, is the primary means of reducing the level of total dissolved solids in the affected aquifer. During this phase, groundwater was extracted from the wellfield and treated using ion exchange and state-of-the-art reverse osmosis technology. These systems removed metals, radionuclides and dissolved solids and generated a high-purity water product (permeate). The permeate was treated with caustic soda or liquid to raise the pH to a level consistent with the restoration target and then re-injected into the affected aquifer. When deemed necessary, a chemical reductant (hydrogen sulfide gas) was added to the injection stream to attenuate trace metal concentrations. The RO permeate injection phase was operated to minimize the consumptive removal of groundwater while still maintaining a hydrologic sink (cone of depression) within the wellfield. During this phase, the wellfield was typically over-recovered by ten to thirty percent, thus continuing the beneficial effect of groundwater sweep from the perimeter of the operating area. The restoration brine water was discharged to two Class I injection wells (WDEQ Permit Number UIC 00-340).

The aquifer recirculation phase was done at the end of restoration and simply involved circulation of the entire wellfield to provide consistent water quality. It was done at the end of the RO phase (the last pore volume) or when reductant (H₂S) gas was added. The recirculation was found to increase oxygen levels to the wellfield and so volumes circulated were limited. Typically the last pore volume (PV) of the RO phase included the addition of reductant and circulation throughout the wellfield. At the end of recirculation and/or H₂S addition, the restoration was typically deemed complete. In the future, it is recommended that

recirculation not be done due to the introduction of oxygen through the circulation process.

During the stabilization phase, the only wellfield activity was well sampling for the monitoring program. The sample results demonstrate that the chemical constituents in the groundwater of the ore zone are in equilibrium with their immediate surroundings. The above phases of restoration were used to treat the affected wellfield groundwater for a specified sequence and volume to reach the restoration goals based on restoration experience.

7.4 RESTORATION VOLUMES

The Restoration and Reclamation Plans in Section 6 of the Permit to Mine No. 478, A2 Update (COGEMA 1996) provided the following treatment volumes for each of the active phases of restoration:

- Groundwater Sweep – one (1) pore volume displacement (PVD) of treatment
 - RO Permeate Injection – five (5) PVDs
 - Recirculation – one (1) PVD
- Total = seven (7) PVDs of treatment

One pore volume (PV), was calculated as follows:

$PV = (WA) (AT) (FF) (P) (CONV)$ where:

PV = pore volume

WA = wellfield area in square feet (ft²)

AT = average thickness in feet (ft)

FF = flare factor of 1.44 [effectively adds 44% to the volume to account for a 20% horizontal flare of lixiviant and a 20% vertical flare of lixiviant]

P = average porosity (26%)

CONV = conversion from ft³ to gallons (7.48)

It is noted that the flare factor of 1.44 is a regulatory requirement from WDEQ. Based on historical operations and data, actual values for horizontal and vertical flare are typically much lower than 20 percent and usually less than 5 percent. Because of the required 1.44 flare factor, COGEMA has processed a volume of water during restoration that greatly exceeds the amount that would be required based on historical operational data.

The length of time each of the restoration methods was used was dependent on the local groundwater conditions of each mine unit at the time of restoration and is described in detail in the RDP attached to this report. The total PVDs during the restoration for each mine unit exceeded the required 7 PVDs to accomplish the goals of the restoration program.

7.5 RESTORATION IMPROVEMENTS

During the restoration process phase, several improvements were made to the restoration process including:

- addition of two RO units to the Satellite Plant to increase the total RO feed capacity to 1,000 gpm;
- installation of two air stripping columns to remove dissolved carbon dioxide gas from the RO permeate stream prior to injection into the wellfield, resulting in pH between 7.5 and 8.0 and TDS around 100 mg/l in the injection stream;
- reduction in overall wastewater stream by 40 to 50% through addition of an another waste water RO unit that further concentrates the RO brine from the primary RO units and,
- recirculation volumes were limited in order to reduce oxygen input into the system, but more PVs of RO were processed instead.

7.6 RESTORATION MONITORING

7.6.1 Water Quality Monitoring During Restoration

The groundwater monitoring plan for the restoration program of Christensen Ranch was provided in Section 5.8 of Permit to Mine 478, A2 Update (COGEMA 1996). The plan was followed for the restoration programs as described below.

7.6.1.1 Ore Zone Water Quality

The progress of groundwater sweep and permeate injection was monitored in each MU by sampling and analyzing the groundwater on a weekly basis as well as on a treated volume basis as follows:

- Weekly samples of the recovery stream composite were taken and analyzed for parameters such as bicarbonate, sulfate, chloride, conductivity, pH, and uranium. Other parameters such as sodium, calcium and TDS were added periodically to assess the progress of restoration. These samples were analyzed by COGEMA's on-site laboratory, or on occasion submitted to an outside commercial laboratory.
- Weekly samples of the recovery stream composite were taken and analyzed for parameters such as bicarbonate, sulfate, chloride, conductivity, pH and uranium.
- Designated restoration wells were sampled at the end of each restoration phase and analyzed for the WDEQ Guideline 8 list of parameters. These samples were analyzed by an outside commercial laboratory. Each designated restoration well was previously utilized as a baseline water quality well.

- The designated restoration wells for each MU were established in the Baseline Water Quality Data Packages or the Wellfield Data Packages for the respective MUs. The MU specific restoration wells are listed in the attached RDPs.

7.6.1.2 Ore Zone Perimeter Monitor Wells

The ore zone perimeter monitor wells were sampled monthly throughout the active restoration phases and quarterly thereafter. Designated ore zone perimeter monitor wells for each MU are listed in the attached RDPs. The samples were analyzed for chloride, conductivity and total alkalinity to determine if the wells had been affected by mining solutions. The monitoring results were reported in the *Irigaray and Christensen Ranch Projects Semi-Annual Reports, WDEQ Permit to Mine 478*.

7.6.1.3 Shallow Monitor Zone

Shallow zone monitoring wells were designated in the Baseline Water Quality Data Packages or the Wellfield Data Packages for each MU. The shallow monitor zone wells were sampled monthly throughout the active restoration phases and quarterly thereafter. The samples were analyzed for chloride, conductivity and total alkalinity to determine if the wells had been affected by mining solutions. The monitoring results were reported in the *Irigaray and Christensen Ranch Projects Semi-Annual Reports, WDEQ Permit to Mine 478*.

7.6.1.4 Deep Monitor Zone

Deep monitor zone wells for each MU were designated in the Baseline Water Quality Data Packages or the Wellfield Data Packages for each MU. The deep zone monitor wells were sampled monthly throughout the active restoration phases and quarterly thereafter. The samples were analyzed for chloride, conductivity and total alkalinity to determine if the wells had been affected by mining solutions. The monitoring results were reported in the *Irigaray and Christensen Ranch Projects Semi-Annual Reports, WDEQ Permit to Mine 478*.

7.6.2 Water Level Monitoring During Restoration

The water level of all monitor wells was measured routinely throughout the life of the operations. Water levels were taken prior to each monitor well sampling event. The water level results, along with potentiometric surface maps of the ore, shallow and deep monitor zones, were presented annually in the *Irigaray and Christensen Ranch Projects Semi-Annual Reports, WDEQ Permit to Mine 478*.

8 ASSESSMENT OF FUTURE GROUNDWATER QUALITY

Following completion of the restoration phase of an ISR project, some groundwater constituents may remain elevated above baseline concentrations. To protect the quality of adjacent groundwater, the ISR operator must assess whether these residual constituents could migrate downgradient past aquifer exemption boundaries at concentrations that would impact the future use of those waters. Key issues to consider include the geochemical conditions in the aquifer prior to ISR mining, the effects of ISR mining and subsequent groundwater restoration on these geochemical conditions, and the likely effects of long-term groundwater flow and natural attenuation on constituent transport from the mined area.

Geochemical conditions in the ore zone aquifer prior to, during, and after mining and restoration control the mobility of groundwater solutes associated with uranium mineralization (Demuth 2006). The primary factor controlling the mobility of solutes associated with roll-front deposits is redox condition. Under the reducing conditions that are naturally present in roll-front deposits, most solutes are strongly attenuated and relatively immobile. Solute are generally more soluble and mobile under oxidizing conditions. ISR operations typically introduce oxidants into the native groundwater to enhance the solubility and mobility of uranium in the mineralized zone. ISR operations also tend to mobilize constituents other than uranium, including radium, iron, manganese, selenium and sulfate.

The re-establishment of long-term reducing conditions in the restored aquifer is an important factor that can serve to limit the migration of constituents affected by ISR mining because reducing conditions have a major effect on the mobility of many constituents associated with uranium roll front deposits, including uranium, selenium, arsenic, molybdenum, and sulfur. These constituents are likely to be relatively immobile under more reducing conditions, although adsorption onto clay and iron oxyhydroxide minerals in the aquifer is likely to cause some attenuation of uranium, selenium, arsenic, and molybdenum, even under oxidizing conditions. Radium-226 attenuation is not directly dependent on redox conditions, and its mobility is instead limited by the formation of solid solutions with other constituents such as barium to form barite [BaSO₄] and by adsorption onto clay minerals in the aquifer (Demuth 2006).

In some cases, relatively low concentrations of mining-related constituents may remain in the groundwater after restoration, or may reappear during groundwater stabilization. However, the offsite migration of these constituents is likely to be limited by the effects of dispersion and geochemical attenuation. Deutsch et al. (1985) reported experimental results indicating that leached ore-zone materials obtained from a Texas in-situ leach site retained significant reducing capacity even after being subjected to lixiviants. In addition, most sites will retain their original groundwater flow directions and the downgradient reduced sediments in the rock matrix have been shown to strongly attenuate these constituents.

8.1 CONSTITUENTS REQUIRING ASSESSMENT

Following application of Best Practicable Technology (BPT) with respect to aquifer restoration of the ISR MUs, several constituents exceeded the TRV and either the Wyoming Class I Standard or the EPA MCL within one or more of the Christensen Ranch MUs. The constituents that exceeded standards include uranium, radium-226, iron, manganese, selenium, and sulfate. TDS also exceeded the Wyoming Class I Standard at MU4. A discussion of the geochemical properties for each of these constituents, except for TDS, which is an aggregate measurement of all the major cations and anions in solution, is presented under the geochemical assessment portion of the transport assessment that follows.

8.2 TRANSPORT ASSESSMENT

This transport assessment evaluates potential migration pathways for constituents regulated under Permit to Mine No. 478 and the U.S. Nuclear Regulatory Commission (NRC) source material license No. SUA-1341. Key components of the assessment are geochemical and hydrologic factors that control solute transport. The hydrologic component defines the rate and direction of groundwater flow within the aquifer. The geochemical component considers the reduction in solute concentrations that occur along groundwater flowpaths. Information specific to the individual MUs are provided in the respective RDPs.

No groundwater modeling specific to the Christensen Ranch MUs is included in this report. Christensen Ranch site conditions, including the constituents of concern, are similar to those at the Irigaray Mine. Groundwater modeling included in the Irigaray Mine Aquifer Restoration Report (COGEMA 2003) is referenced where applicable to the Christensen Ranch Site.

8.2.1 Geochemical Assessment

The geochemical component of the transport assessment addresses the physical and chemical behavior of constituents of concern under the prevailing environmental conditions at the site. Much of the information provided in this report on the geochemical properties of the constituents of concern has been summarized in a report prepared for Uranium Resources, Inc. titled Fate and Transport of Groundwater Constituents at In-Situ Uranium Leach Facilities (Demuth 2006).

8.2.1.1 Uranium

Redox conditions strongly influence the groundwater mobility of uranium. Under reducing conditions, uranium(IV) forms relatively insoluble solids such as uraninite and coffinite, minerals that are typically found in uranium roll-front deposits. In roll-front deposits, uranium is concentrated in zones close to or in

contact with the interface between the oxidized and reduced sandstones (Harshman 1974). At pH values relevant to most groundwater systems, low concentrations of dissolved uranium(IV) are present in solution mostly as hydrolysis species $[U(OH)_3^+$ and $U(OH)_4^0]$, because uranium(IV) is typically not strongly complexed by anions such as chloride, fluoride, phosphate, or sulfate (Langmuir 1978, EPA 1999).

Uranium exists in the +VI oxidation state in oxygenated groundwaters. Common uranium(VI) mineral phases include schoepite, rutherfordine, uranophane, tyuyamunite, carnotite, autunite, and potassium autunite (Langmuir 1978, EPA 1999). Uranium(VI) minerals tend to be relatively soluble, so uranium(VI) concentrations in oxygenated groundwaters are usually controlled by adsorption rather than by precipitation of uranium(VI) solids (EPA 1999). At pH values observed in most groundwater systems, uranium-carbonate aqueous species are likely to make up the majority of dissolved uranium(VI), including $UO_2CO_3^0$, $UO_2(CO_3)_2^{2-}$, and $UO_2(CO_3)_3^{4-}$. Solution pH and carbonate concentrations are the most important factors influencing the adsorption behavior of U(VI). The relatively high solubilities of uranium(VI) solids plus the ability of carbonate complexes to increase uranium(VI) solubility are used during uranium in-situ recovery to maximize the amounts of uranium dissolved in the lixiviant.

Aqueous uranium and its complexes sorb onto clays, organics, and iron oxides (EPA 1999). Uranium sorption by soils generally reaches a maximum in the pH range from pH 5 to 8 (EPA 1999). Higher-ionic-strength solutions or the presence of carbonate ions tend to decrease uranium(VI) sorption. Uranium can also be attenuated in groundwater through coprecipitation reactions with metal hydroxides such as iron hydroxide.

An extensive body of literature exists regarding partition coefficients (K_d) for uranium. Various studies have evaluated partition coefficients as a function of pH (Erickson 1993, Gilbin 1980, and Serkiz 1994), ionic strength (Kaplan 1998), solvent composition (Salter 1981, and Warnecke 1988), dissolved carbonate concentration (Hsi and Langmuir 1985, and Tripathi 1984), grain size (Serkiz 1994, and Serne 1993), mineralogy (Ames 1982, Borovec 1981, and Waite 1992) and saturation conditions (Lindenmeier 1995, and Kaplan 1995). Results of these and other studies indicate a range of over 6 orders of magnitude for K_d values of uranium. Representative published compilations of K_d values include Baes and Sharp (1983), Looney et al (1987), Thibault et al (1990) and McKinley and Scholtis (1993). Baes and Sharp list 24 uranium K_d values ranging from 10.5 to 4,400 ml/g for agricultural silts and clays in the pH range of 4.5 to 9.0 and provide an estimated default value of 45 ml/g. Looney et al (1987) in a study of geochemical parameters needed for environmental assessments at the DOE Savannah River Plant, list a "recommended" K_d of 39.8 ml/g but a range from 0.1 to 1,000,000 ml/g. Thibault et al. indicated a range of 0.03 to 395,100 ml/g for 46 samples. However the geometric mean uranium K_d by soil type was 35 ml/g for sand, 15 ml/g for loam, 1,600 ml/g for clay and 410 ml/g for organics.

The general trend in uranium K_d values as a function of pH is that adsorption is low at pH values of 3 s.u. or less, increases rapidly from pH of 3 to 5, reaches a maximum between pH of 5 and 8 and then decreases with increasing pH greater than 8 s.u. The decrease in adsorption at high pH is actually related to the presence of dissolved carbonate. At near- and above-neutral pH conditions, dissolved U(VI) forms strong anionic uranyl-carbonate complexes with dissolved carbonate, making it less likely to adsorb to the surface-charged soil minerals (Tripathi 1984, Hsi and Langmuir 1985, and Waite 1992). Soils containing larger percentages of iron oxide minerals and mineral coatings and/or clay minerals will exhibit higher sorption characteristics for uranium than soils dominated by quartz and feldspar minerals (Waite 1992).

The EPA (2004) provides a uranium K_d look-up table based on pH values. Ambient and post-restoration pH for groundwater within the Christensen Ranch Mine Units is generally between 7.5 and 9.0 s.u. The EPA look-up table indicates that, for pH between 7 and 9, the minimum K_d range is from 0.4 to 63 ml/g and the maximum value range is from 7,900 to 630,000 ml/g.

8.2.1.2 Radium

Radium, an alkaline earth element, is generally relatively immobile but can be mobilized under some conditions. Ra-226 concentrations in groundwater can vary from 0 to over 200 pCi/l, but most values are less than 10 pCi/L (Langmuir 1985). Radium-226 and radium-228 are present in uranium roll-front deposits because of the decay of uranium-238 and thorium-232, respectively. Groundwater radium concentrations commonly are elevated in the ore zone relative to the background levels present immediately upgradient and downgradient of the ore (Wanty et al. 1987). Radium is present in natural waters only as the Ra^{2+} ion and its complexes. Radium concentrations in natural waters are usually controlled by adsorption and/or precipitation via solid-solution (Langmuir 1985). Radium is frequently precipitated from sulfate-rich waters as a solid solution with barium (as barite, $BaSO_4$) or less commonly with calcium (as gypsum, $CaSO_4 \cdot 2H_2O$) or strontium (as celestite, $SrSO_4$) (Langmuir and Melchior 1985, Langmuir and Riese 1985). The $(Ra,Ba)SO_4$ coprecipitation reaction has been shown to be an important process in controlling radium solubility in natural waters (Pardue 1998).

In general, radium adsorption on mineral surfaces at low pH is minimal, increasing with increasing pH. For iron oxides, the increase in adsorption begins around pH of 6 to 8 and reaches a maximum of around 10 s.u. or less (USEPA 2004). Radium can be attenuated by adsorption onto clays (Langmuir and Chatham 1980, Kaufmann et al. 1976, Granger 1963). Radium adsorption decreases with increasing ionic strength because of the effects of competing cations (USEPA 2004). Radium is also strongly adsorbed to mineral oxides, especially at near neutral and alkaline pH conditions (USEPA 2004). Results of

some studies indicate that radium may be strongly adsorbed by organic materials in soils (USEPA 2004).

Compared to most other radionuclides, very limited data are available on radium sorption, particularly K_d values. Moreover, the EPA (2004) states that any data indicating high radium adsorption on geologic materials should be viewed cautiously as $(\text{Ba,Ra})\text{SO}_4$ coprecipitation may have occurred during the measurements. However, from an attenuation standpoint, the amount of radium that is removed from groundwater by adsorption versus precipitation is largely irrelevant.

Serne (1994) measured K_d values in sandy arid soils ranging from 214 to 467 ml/g. The experiments were conducted with pH values slightly below 8 (which are similar to the values observed at Christensen Ranch before and after ISR mining). Looney et al (1987) provided a compilation of K_d values from several sources with a range of 10 to 1,000,000 ml/g and a "recommended" value of 100 ml/g. The "recommended" value was specific to the Savannah River site. Thibault et al (1990) published a compilation of K_d values for four sediment types that ranged from 57 to 530,000 ml/g. Geometric means for sand, silt and clay of 500, 36,000 and 9,100 ml/g were presented. One value for organics of 2,400 ml/g was also included in the compilation. Sakamoto et al. (2001) measured radium-226 K_d values on loam (12,000 to 28,000 ml/g), sand (190 to 2,300 ml/g) and tuff (1,900 to 3,000 ml/g) in dilute solutions with pH values that ranged from 5.7 to 8.0.

Radium K_d values are generally much higher than those for uranium or any of the other constituents of concern at Christensen Ranch. Groundwater modeling performed for the Irigaray Ranch ISR Uranium Mine, located a few miles to the northwest, conservatively used a K_d of 5 ml/g. Results of that modeling were included in the Irigaray Aquifer Restoration Report submitted to WDEQ (COGEMA 2003). The model simulations indicated that residual concentrations of radium-226 would not migrate outside of the aquifer exemption boundary at levels above regulatory standards within the time frame of the simulations (1000 years).

8.2.1.3 Iron

Iron is present as iron(III) oxides (hematite and goethite) and clays on the oxidized side of uranium roll fronts. Iron sulfides, along with organic carbon debris, define the redox conditions on the reduced side of the redox interface within a roll-front deposit and are important reductants for dissolved uranium (Reynolds and Goldhaber 1983). Within the ore zone and downgradient into the unaltered sandstone, generally about half of the iron is present as reduced iron [iron(II)] in the form of pyrite and marcasite [FeS_2] (Harshman 1972, Goldhaber et al. 1978, Reynolds and Goldhaber 1983). The remaining half of the iron is usually present in iron(II)-containing silicates (Harshman 1972). Total iron contents in the ore zone generally range from 1 to 2 wt. % compared to about 0.8 to 1 wt % in the altered and unaltered sandstone.

Iron concentrations in groundwater are typically controlled by the solubility of iron-containing solid phases rather than by adsorption (Rai et al. 1987). Because ferrous iron is relatively more soluble and mobile than ferric iron, redox conditions tend to control groundwater iron concentrations. Dissolved iron is most commonly present in groundwater as the ferrous ion Fe^{2+} . Ferric iron can occur in acid solutions as Fe^{3+} , FeOH^{2+} and $\text{Fe}(\text{OH})_2^+$. However, above a pH of approximately 5, ferric iron (Fe^{3+}) forms relatively insoluble iron(III) oxides and hydroxides, such as amorphous $\text{Fe}(\text{OH})_3(\text{s})$. Amorphous $\text{Fe}(\text{OH})_3(\text{s})$ or other ferric iron solids that form under oxidizing conditions can sorb or coprecipitate trace metals, such as uranium, thereby exerting an important control on groundwater mobility of trace metal constituents (Langmuir et al. 2004). In extremely reducing conditions in the presence of sulfide, iron forms relatively insoluble iron sulfides such as pyrite [FeS_2]

8.2.1.4 Manganese

Manganese tends to be relatively concentrated in uranium roll-front deposits near the redox interface, and appears to be strongly associated with carbonate (Goldhaber and Reynolds 1977). The geochemistry of manganese is similar to that of iron. The mobility of manganese in groundwater is strongly affected by redox conditions. Under typical groundwater conditions, dissolved manganese will usually be present in reduced form as Mn^{2+} . Manganese(II) solid phases that may form include rhodochrosite [MnCO_3], and rarely, $\text{MnS}(\text{s})$ (Langmuir et al. 2004). Under oxidizing conditions, manganese forms relatively insoluble manganese (III) and (IV) oxide phases that are stable above approximately pH 6 to 7 (Rai et al. 1987, Langmuir et al. 2004). These manganese oxides readily adsorb trace metals such as cobalt, zinc, and nickel and can therefore significantly influence trace metal migration in groundwater (Murray 1975, Langmuir et al. 2004).

8.2.1.5 Selenium

Selenium concentrations in uranium roll-front deposits are elevated in narrow zones at the outer edges of the oxidized sandstone, at the redox interface, or in reduced sandstone close to the interface (Harshman 1974). Selenium in uranium roll-front deposits typically occurs in reduced form as the relatively insoluble phases native selenium and ferroselite (FeSe_2) (Harshman 1974).

Selenium exhibits relatively complex redox behavior similar to that of sulfur (Langmuir et al. 2004). In oxidizing groundwater, selenium typically is present as selenite (IV) or selenate (VI) species. Selenite predominates in acidic oxidizing solutions, whereas selenate is the dominant oxidation state in more alkaline oxidizing solutions (McLean and Bledsoe 1992). Above approximately pH 7, selenite and selenate exist primarily as the oxyanions SeO_3^{2-} and SeO_4^{2-} , respectively (Rai et al. 1987).

Under reducing conditions, selenium is typically present as native selenium or in the selenide (-II) oxidation state. Selenide is generally present in solution in the

form of HSe^- above pH 4 (Rai et al. 1987). Native selenium is relatively insoluble, and selenide often forms insoluble metal selenide phases such as ferroselite $[\text{FeSe}_2]$ (McLean and Bledsoe 1992, Langmuir et al. 2004). The insolubility of the selenide minerals is the primary attenuation mechanism for selenium in aquifers hosting roll-front uranium deposits.

Oxidized selenate and selenite species can be reduced through interaction with sediments and removed from solution by binding with organic material or by precipitation (White et al. 1991, Zhiang and Moore 1997). These selenium reduction reactions are generally microbially mediated (Oremland et al. 1989, White et al. 1991). In oxidizing environments, attenuation of selenite and selenate generally occurs through adsorption. Because selenite and selenate ions are anionic, adsorption of these species is greatest at low pH (Neal et al. 1987, Rai et al. 1987, McLean and Bledsoe 1992). Selenite is generally more strongly adsorbed than selenate (Ahrlichs and Hossner 1987, Balistrieri and Chao 1987, Rai et al. 1987, Balistrieri and Chao 1990). Selenite is adsorbed by iron and aluminum oxyhydroxides and by clay minerals such as kaolinite and montmorillonite (Rai et al. 1987, McLean and Bledsoe 1992). Selenate adsorption may occur at lower pH values on iron oxyhydroxides and kaolinite (McLean and Bledsoe 1992). Selenite and selenate adsorption is decreased by the presence of phosphate or sulfate in solution (Balistrieri and Chao 1987, Rai et al. 1987, Balistrieri and Chao 1990, McLean and Bledsoe 1992). Selenite can be oxidized to selenate by reaction with manganese oxide solids (Scott and Morgan 1996).

Selenium K_d values have been summarized by Sheppard and Thibault (1990). They reported geometric mean K_d values of 150 ml/g for sand, 500 ml/g for loam, 740 ml/g for clay, and 1,800 ml/g for organic soils. Langmuir et al. (2004) reported K_d values for selenate ion that varied with pH: 18 ml/g at pH 4.9, 5.0 ml/g at pH 6.8, and 2.2 ml/g at pH 8.0. These lower reported K_d values are likely caused by the relatively poor sorption of the selenate ion compared to the selenite ion. Lee and Murarka (2005) found that selenium K_d values for soils varied with selenium concentration. At an initial solution concentration of 1 mg/L, selenium K_d values ranged from 5 to about 500 ml/g for selenite, and from 2 to 18 ml/g for selenate for the soils tested. The presence of sulfate decreased adsorption of selenium with the greatest effects observed for selenite.

8.2.1.6 Sulfate

Sulfate sulfur appears to be leached from oxidized sandstone in uranium roll-front deposits, where it was initially present as gypsum (Harshman 1974). Higher concentrations of sulfate are observed in the reduced sandstone, possibly as the result of iron(II)-sulfate precipitation during ore formation (Harshman 1974). Sulfur also occurs as sulfide primarily in the form of iron sulfides in the ore zone and unaltered sandstone where reducing conditions exist.

The chemical behavior of sulfur is strongly related to redox properties of the aqueous systems. The sulfate anion (SO_4^{2-}) is present under oxidizing conditions

and is the dominant aqueous form of sulfur under most conditions. Reduced sulfur ion, S^{2-} , is found in groundwater only under highly reducing conditions and forms sulfides of low solubility with many metals (Langmuir et al. 2004).

Sulfate ion is relatively mobile in groundwater, and in many aqueous systems gypsum [$CaSO_4 \cdot 2H_2O$] solubility is the only control on sulfate concentrations (Langmuir 1997). Precipitation of sulfate solids, such as barite [$BaSO_4$] and, less commonly, gypsum, can control radium groundwater concentrations via substitution of the Ra^{2+} ion for Ba^{2+} or Ca^{2+} (Langmuir and Melchior 1985, Langmuir and Riese 1985).

Reduced sulfur in groundwater, predominantly hydrogen sulfide (H_2S) and bisulfide (HS^-), are typically produced by bacterial reduction of sulfate under anaerobic conditions (Chapelle 1993). Many metals, such as iron, copper, nickel, and zinc, form insoluble sulfide solids under reducing conditions. Pyrite [FeS_2] is abundant in the reduced portion of most uranium roll-front deposits (Harshman 1974)

8.2.2 Hydrologic Assessment

In addition to the geochemical assessment, the effects of long-term groundwater flow, including advective mixing and dispersion, on constituent concentrations is considered under the transport assessment. Direction and velocity of groundwater flow are critical hydrologic factors with respect to solute transport. Determination of groundwater flow direction is based on water-level data that were routinely collected from the monitor well networks for each of the MUs. The velocity of groundwater flow depends on hydraulic conductivity, hydraulic gradient, and porosity. Darcy's Law is the governing equation used to calculate groundwater velocity and is described as follows:

$$v = k i / \theta \quad \text{where}$$

v is groundwater velocity
 k = hydraulic conductivity
 i = hydraulic gradient
 θ = effective porosity

The hydraulic conductivity and porosity are intrinsic properties of the aquifer matrix and do not vary with time. Hydraulic conductivity estimates are derived from pumping tests. Porosity estimates are derived from core data and literature. Hydraulic gradient varies in time and space depending on changes to the groundwater flow regime. Under normal non-stressed conditions, changes in hydraulic gradients within an aquifer tend to be minor and occur gradually. When a groundwater flow regime is stressed (because of extraction, injection, mine dewatering, mounding from seepage, etc.) hydraulic gradients may change abruptly, resulting in measurable changes in groundwater flow velocity and direction.

8.2.2.1 Groundwater Flowpaths and Travel Times

Groundwater within the uranium producing aquifer (the K Sandstone) flows predominately to the west-northwest. Data specific to each of the MUs are presented in detail under the respective MU RDPs. The 'K' Sandstone is a confined aquifer, bounded above and below by areally extensive claystones, siltstones and shales, as described in Section 5. Large water level differences between the 'K' Sandstone and the overlying 'J' Sandstone and underlying 'L' Sandstone confirm that these units are not in hydraulic communication. Therefore, groundwater within the 'K' Sandstone is primarily restricted to horizontal flow, with minimal movement, if any, into the overlying or underlying aquifers. There are no known surface discharge points for groundwater from the 'K' Sandstone with several miles of the site. The closest possible surface discharge point for the 'K' Sandstone would be the Powder River, six to eight miles to the northwest.

Groundwater velocities were calculated for each of the MUs based on hydraulic conductivity, natural hydraulic gradient and porosity estimates. Travel times were calculated for the time to reach the monitor ring (typically 400 feet from the wellfield edge) and the Permit Boundary. The MU RDPs present the data used in the calculations. The range of groundwater velocity determined from those calculations was from 0.0088 to 0.043 ft/d (3.2 to 15.5 ft/yr). Estimated travel times to reach the 400 foot monitor well ring ranged from 26 to 123 years. The minimum distance to the permit boundary varies from one MU to the next, ranging from 2,500 to 8,500 feet. Under a natural hydraulic gradient, travel times for groundwater from a MU to reach the Permit Boundary ranged from 160 to 2,600 years.

8.2.2.2 Potential Receptors

As previously described, there are no surface water discharge points for 'K' Sandstone groundwater within several miles of the site. Therefore, the only possible receptors of 'K' Sandstone groundwater migrating from the site would be from wells completed within or through that unit. Groundwater rights within and around the permit area are predominately for stock wells. Within the Permit Boundary there are two domestic wells (P24096P, P24085P), one stock/domestic well (P28847W) and one miscellaneous well (P40282W) and several stock wells. Immediately northwest of the permit area, in the direction of groundwater flow there is one domestic/stock well (P28846W), one miscellaneous well (P30346W), and one industrial well (P52981W) and numerous stock wells. The locations of the groundwater rights in the vicinity of the Christensen Ranch Permit Area are shown in Figure D.6.3.1 of the Christensen Ranch Amendment Application To WDEQ Permit to Mine No. 478. Appendix A includes a listing of all known groundwater rights in Townships 44 and 45 North, and Ranges 76 and 77 West.

8.2.3 Fate and Transport of Monitored Constituents

As was previously stated, no groundwater modeling was performed specifically for the Christensen Ranch site. However, extensive groundwater flow and solute transport modeling was performed for the Irigaray Mine site located six to ten miles northwest of Christensen Ranch. Those modeling results were included in the Irigaray Aquifer Restoration Report and accepted by WDEQ and the NRC and are applicable in a general sense to Christensen Ranch.

The model focused on the impacts of advective mixing on constituent concentrations migrating from the Irigaray site but did not address the effects of geochemical processes along the flowpath. The results of the model indicated the maximum concentration at a distance of 400 feet from the wellfield was approximately 7 times lower than the initial average wellfield concentration, after subtracting out the average background concentration. The background concentration was subtracted from the impacted concentrations to normalize the data. The monitor well ring is located 400 feet from the wellfield. Calculation of the reduction factor for select constituents is presented below.

Table 8.1 Calculation of the Reduction Factor Determined From Modeling Advective Mixing of Groundwater Over a Distance of 400 feet (Irigaray Wellfield Restoration Report)

Constituent	Post Restoration Average Wellfield Concentration (mg/l) (A)	Background Concentration (mg/l) (B)	Maximum Concentration @ 400 ft (mg/l) (C)	Reduction Factor $\frac{(A-B)}{(C-B)} = R$
Manganese	0.18	0.0106	0.034	7.23
Selenium	0.04	0.0026	0.008	6.93
TDS	650	379	420	6.60

Geologic and hydrologic conditions are very similar between the sites. The Wasatch Formation is the ore-bearing unit at both sites. Hydraulic conductivity, hydraulic gradient and porosity are similar between the two locations. It is anticipated that groundwater modeling for the Christensen Ranch site would show a similar degree of concentration reduction (6 to 8 times) as demonstrated at the Irigaray Mine site. A reduction factor of 6 was applied to the residual post-restoration wellfield concentrations for constituents that exceeded either Wyoming Class I Standards or EPA MCLs within specific MUs. The calculations are presented in the MU RDPs.

9 SUMMARY AND CONCLUSIONS

COGEMA has prepared this document as a technical demonstration supporting completion of the wellfield restoration operations at MU2, MU3, MU4, MU5 and MU6 of the Christensen Ranch in-situ uranium mine in Johnson and Campbell Counties, Wyoming. Many of the conclusions presented in this section are based on the MU specific data presented in each of the RDPs

Wellfield restoration operations were initiated in MU2, MU3, and MU4 in 1997 and in MU5 and MU6 in 2000. Restoration of all MUs, including stability monitoring, was completed by 2006. An average of 10.1 pore volumes of water were treated for the five Christensen Ranch MUs during restoration activities. Groundwater within the production zone has been restored to the pre-mining class of use, using BPT, as required by the WDEQ.

A summary of the restoration results is presented in the following table

Table 9.1 Restoration Results by Mine Unit, Christensen Ranch

Category	MU2	MU3	MU4	MU5	MU6
Total No. of Monitored Parameters	35	35	35	35	35
No. of Parameters < TRV	24	27	21	25	27
No. of Parameters > TRV but with no WDEQ Class I or EPA MCL	5	5	6	4	4
No. of Parameters > TRV but < WDEQ Class I or EPA MCL	2	2	2	3	1
No. of Parameters > TRV and > WDEQ Class I or EPA MCL	4	1	6	3	3
Parameters > TRV and > WDEQ Class I or EPA MCL	Fe, Mn, U, Ra-226	Mn	Fe, Mn, Se, U, Ra-226, TDS	Mn, Se, U	Mn, Se, U

TRV – Target Restoration Value

WDEQ Class I-Drinking water use

EPA MCL – Maximum Contaminant Limit

For most constituents, restoration activities resulted in a 70 to 90 percent reduction in mean concentrations compared to post-mining levels. Manganese,

selenium, uranium and radium-226 most commonly exceeded the TRVs and Wyoming Class I Standard or EPA MCL.

Manganese was reduced by an average of nearly 70 percent from post mining concentration in the five MUs as a result of restoration activities. Advective mixing of groundwater and adsorption processes should lower the concentration to below the Wyoming Class I Standard and the EPA MCL within relatively short distances from the wellfield.

Selenium was reduced by over 90 percent from post-mining concentrations in all of the MUs except for MU5 where the reduction was only 27 percent. Under reducing conditions, selenium is typically present as native selenium or in the selenide (-II) oxidation state. Native selenium is relatively insoluble, and selenide often forms insoluble metal selenide phases. The insolubility of the selenide minerals is the primary attenuation mechanism for selenium in aquifers hosting roll-front uranium deposits. Advection coupled with geochemical processes should lower the concentration to below the Wyoming Class I Standard and the EPA MCL within relatively short distances from the wellfield.

Uranium was reduced by an average of 90 percent from post mining concentration in the five MUs. However the post-restoration mean concentration is still greater than the recently revised EPA MCL of 0.03 mg/l at each MU. It should be noted that during active restoration the EPA MCL and Wyoming Class I Standard was 5.0 mg/l and the final restoration quality would have met those standards at each of the MUs. Significant attenuation of uranium will occur as groundwater from the wellfields moves into the downgradient reducing portions of the aquifer.

Radium-226 was not as responsive to restoration actions as many of the other constituents but post-mining concentration was reduced by 50 percent or more at all MUs except MU2. The post restoration radium-226 levels were generally close to the TRVs. Although radium-226 does not meet either Wyoming Class I Standard or EPA MCLs, the TRV was also significantly higher than those standards. This is an indication that the groundwater within the MUs was not suitable for drinking prior to mining activities. Elevated radium-226 is consistent with pre-mining water quality and class of use for the site. Furthermore, radium-226 is a natural constituent of the orebody and is relatively immobile in groundwater systems and will not migrate appreciably from the site.

The results of the final water quality during stability monitoring demonstrate that the Christensen Ranch MU2 through MU6 have been successfully restored using best practicable technology. Although all constituents were not restored to background concentrations, or to Class I standards, the groundwater was clearly returned to a quality of use that is equal to or better than, and consistent with the uses for which the water was suitable prior to the operation (Class IV or V). Further restoration at this site is not deemed justifiable in terms of the technical

practicability or economic reasonableness of reducing the chemical concentrations any further. Current conditions are likely to result in continued concentration decreases for primary constituents of concern. Based on the restoration work conducted and discussed herein, COGEMA considers that restoration consistent with Permit No. 478 has been completed, and requests that restoration approval be granted.

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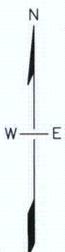
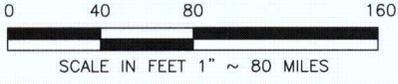
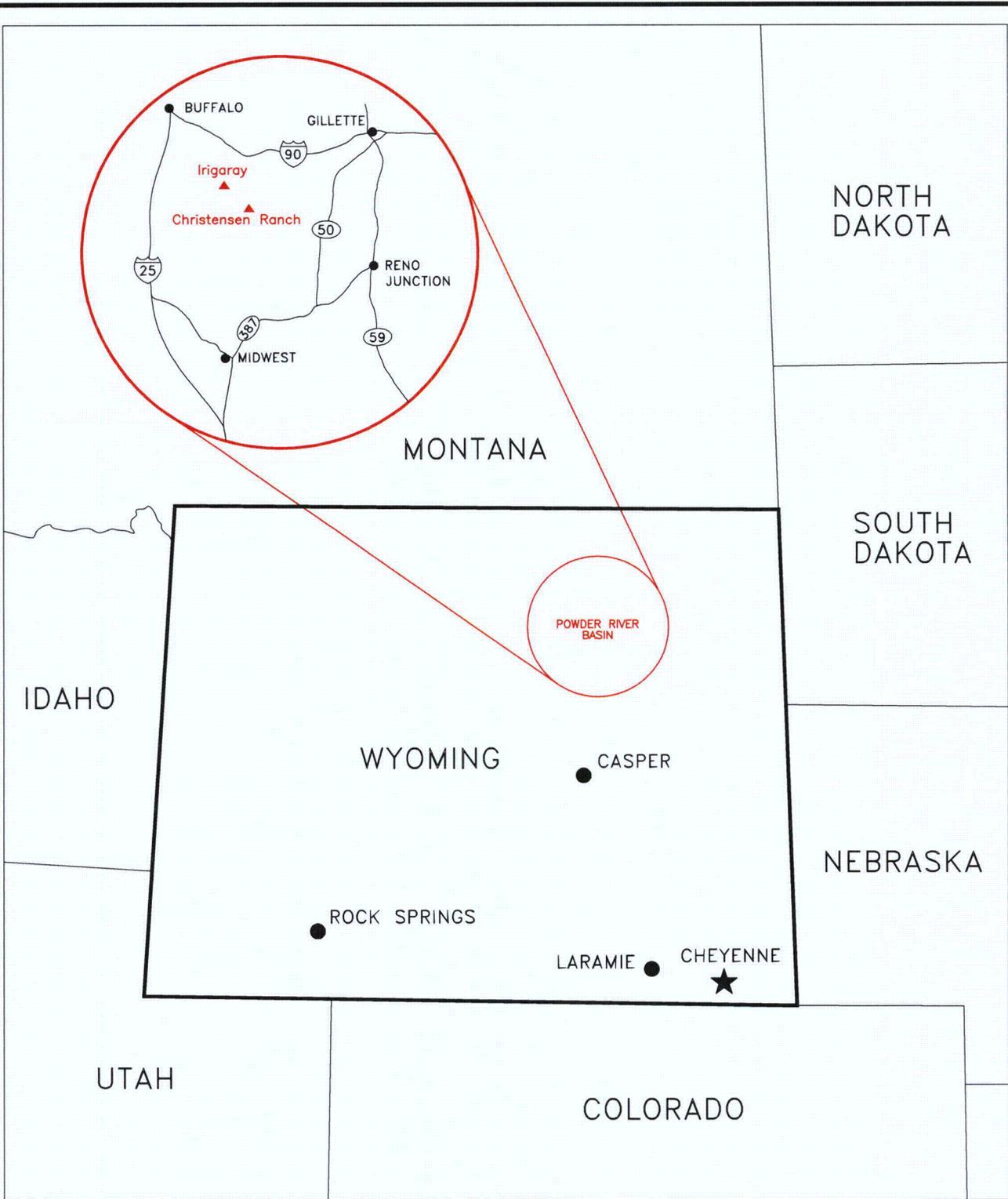
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TABLES

Table 7-1 Target Restoration Values for Christensen Ranch ISR Mine Units

Parameter	Mine Unit 2	Mine Unit 3	Mine Unit 4	Mine Unit 5	Mine Unit 6
Major Cations/Anions					
Ca (mg/l)	10.25	26.3	14.5	24.1	50.1
Mg (mg/l)	0.77	2.1	1.64	3.7	10.8
Na (mg/l)	164.4	152.3	151.7	191.7	278.5
K (mg/l)	5.1	8.4	10.3	19.9	15.3
CO3 (mg/l)	24.0	28.5	24.1	61.9	18.2
HCO3 (mg/l)	205.1	170.7	152.9	217.6	138.3
SO4 (mg/l)	301.6	229.1	238.2	348.2	680.5
Cl (mg/l)	10.1	11.6	11.1	11.9	8.6
NH4 (mg/l)	0.12	0.17	1.20	0.72	0.27
NO2 (N) (mg/l)	0.10	0.04	0.10	0.10	0.50
NO3 (N) (mg/l)	0.12	0.19	0.21	16.8	0.31
F (mg/l)	0.24	0.35	0.30	0.31	0.34
SiO2 (mg/l)	7.9	12.8	12.7	12.3	10.80
General Water Quality Parameters					
TDS (mg/l)	808.8	482.8	508.2	619.3	1160.8
Cond. (umho/cm)	1365.4	775.3	802.7	1095.0	1697.7
Alk. (as CaCO3 (mg/l))	121.4	121.6	119.0	158.4	106.6
pH (units)	9.1	10.42	10.05	10.16	13.60
Trace Metals					
Al (mg/l)	0.1	0.1	0.1	0.10	0.30
As (mg/l)	0.002	0.011	0.007	0.006	0.01
Ba (mg/l)	0.1	0.10	0.1	0.10	0.05
B (mg/l)	0.1	0.10	0.1	0.10	0.10
Cd (mg/l)	0.01	0.01	0.01	0.01	0.05
Cr (mg/l)	0.05	0.05	0.05	0.05	1.54
Cu (mg/l)	0.01	0.077	0.01	0.10	0.05
Fe (mg/l)	0.05	3.051	0.05	0.05	0.81
Pb (mg/l)	0.05	0.05	0.05	0.05	0.01
Mn (mg/l)	0.01	0.01	0.01	0.01	0.05
Hg (mg/l)	0.001	0.001	0.001	0.001	0.002
Mo (mg/l)	0.1	0.1	0.1	0.10	0.14
Ni (mg/l)	0.05	0.05	0.050	0.05	0.26
Se (mg/l)	0.003	0.029	0.010	0.026	0.02
V (mg/l)	0.1	9.22	0.1	0.14	0.78
Zn (mg/l)	0.01	0.10	0.228	0.107	0.04
Radionuclides					
U (mg/l)	0.034	0.376	0.23	0.076	0.055
Ra 226 (pCi/l)	214	270.79	83.0	289.8	428.5

FIGURES

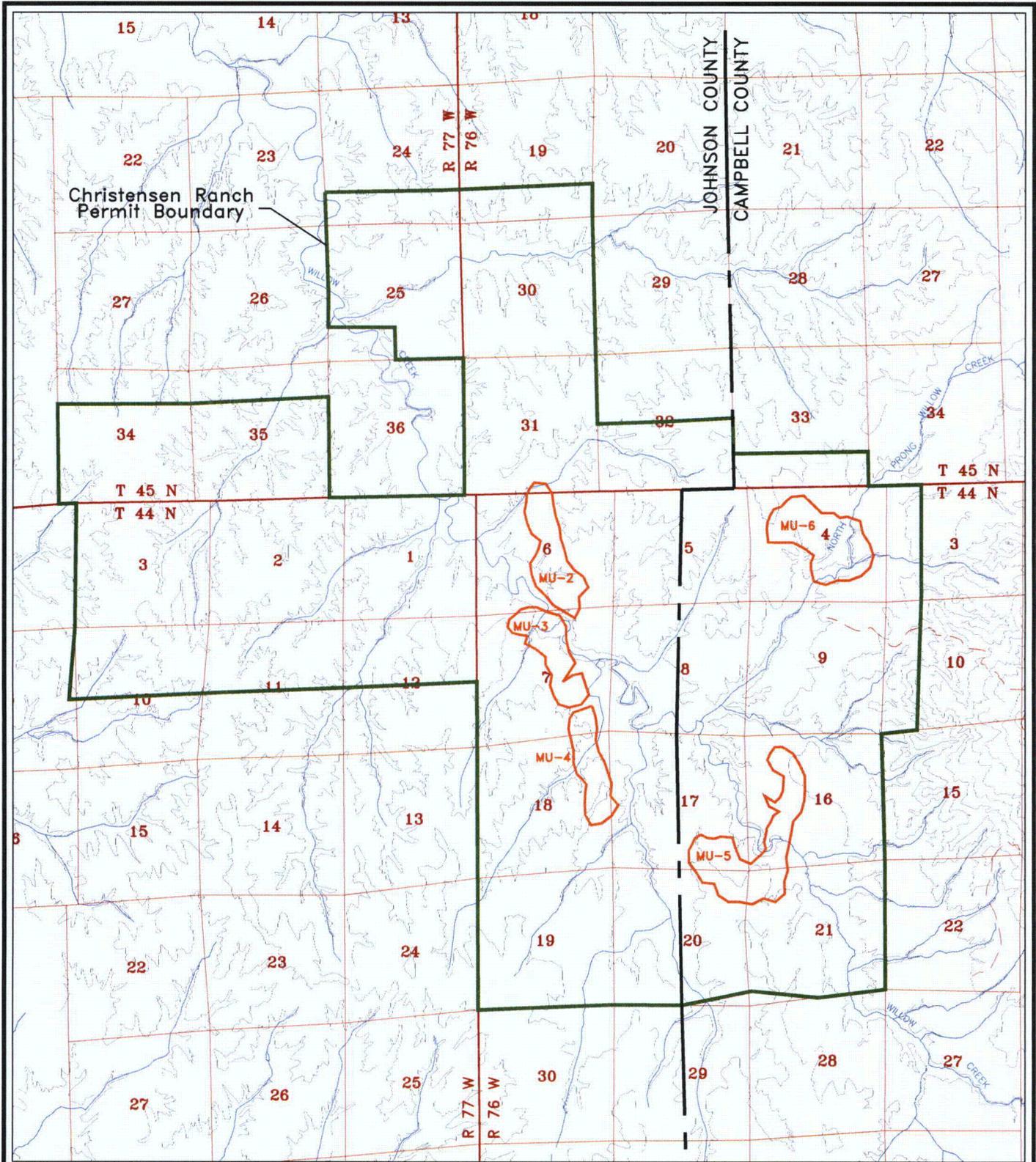


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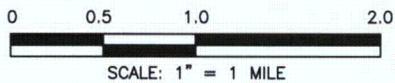
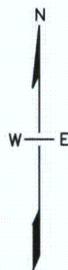
FIGURE 1-1
CHRISTENSEN RANCH AND IRIGARAY MINES
SITE LOCATION MAP

PROJECT: 217-31	DATE: APRIL 2007
CMI MUR Fig 1-1	BY: KRS CHECKED: EL

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LEGEND



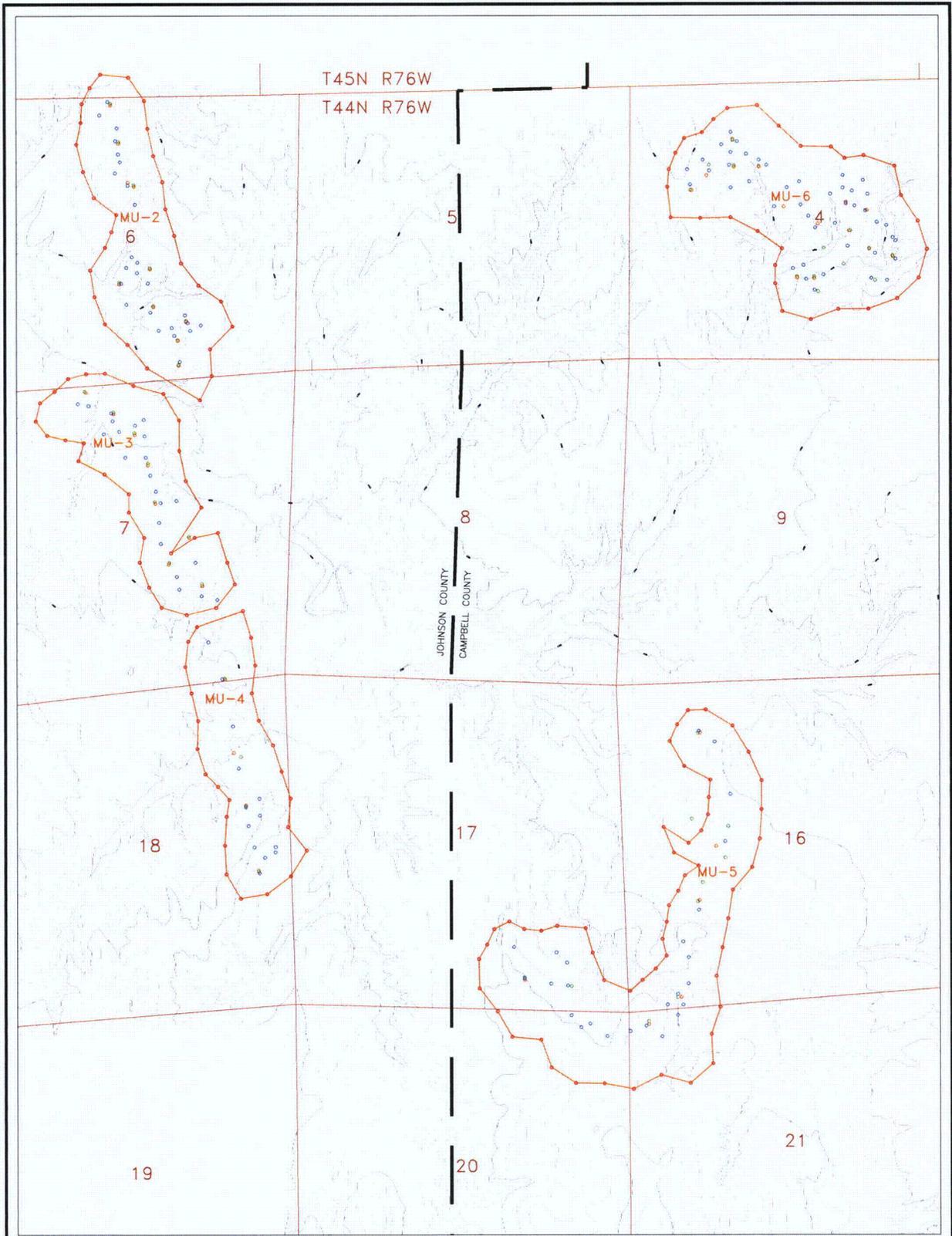
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FIGURE 3-1
CHRISTENSEN RANCH PROJECT
PROJECT LOCATION MAP

PROJECT: 217-12	DATE: APRIL 2007
CMIMURCOMPOSITE.dwg	BY: KRS CHECKED: EL

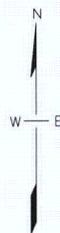
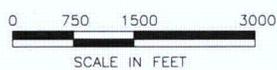
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LEGEND

- PERIMETER MONITORING WELLS
- SHALLOW MONITORING WELLS
- DEEP MONITORING WELLS
- RESTORATION WELLS



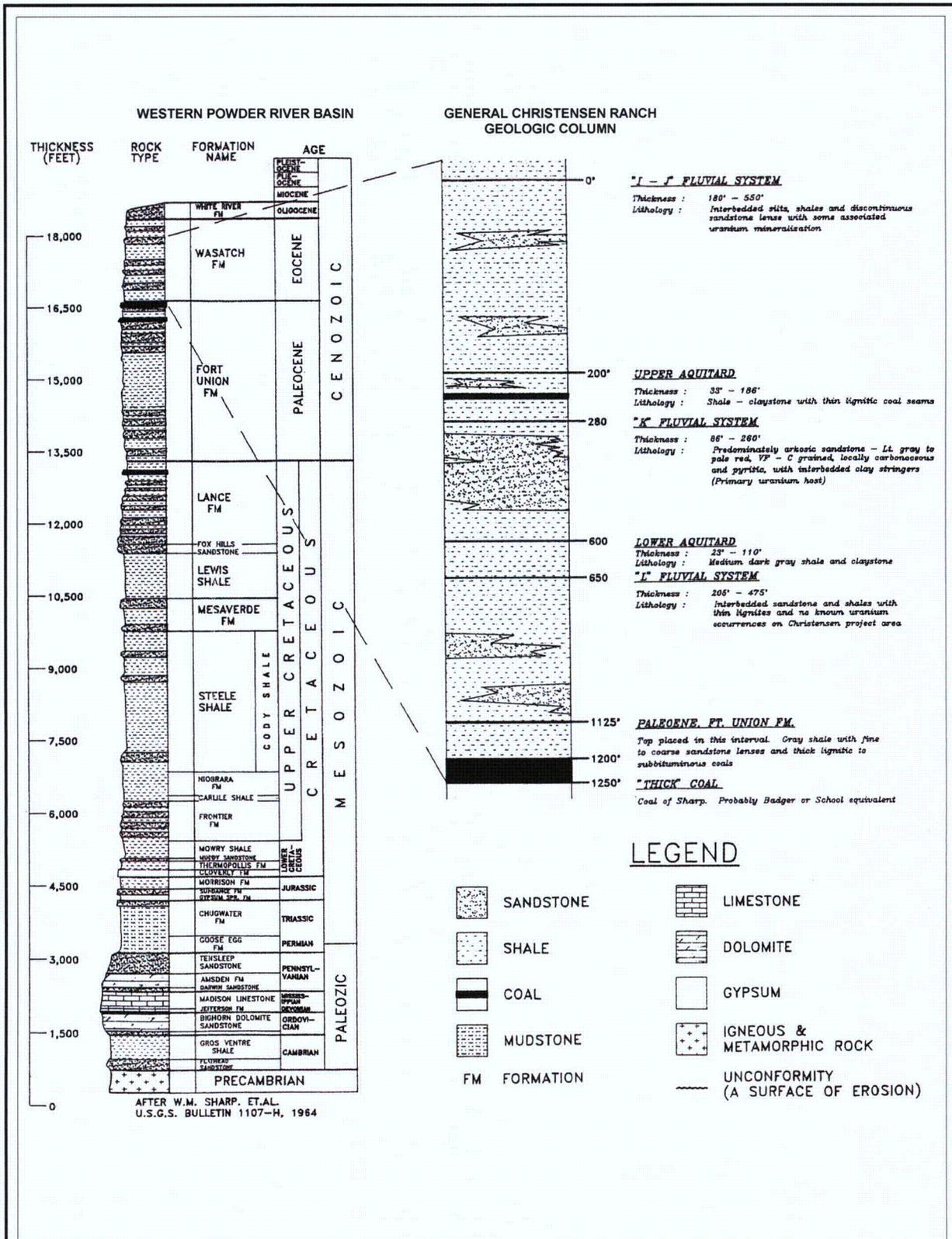
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FIGURE 3-2
CHRISTENSEN RANCH PROJECT
MINE UNIT LOCATION MAP

PROJECT: 217-12	DATE: APRIL 2007
CMI Restoration.dwg	BY: KRS CHECKED: EL

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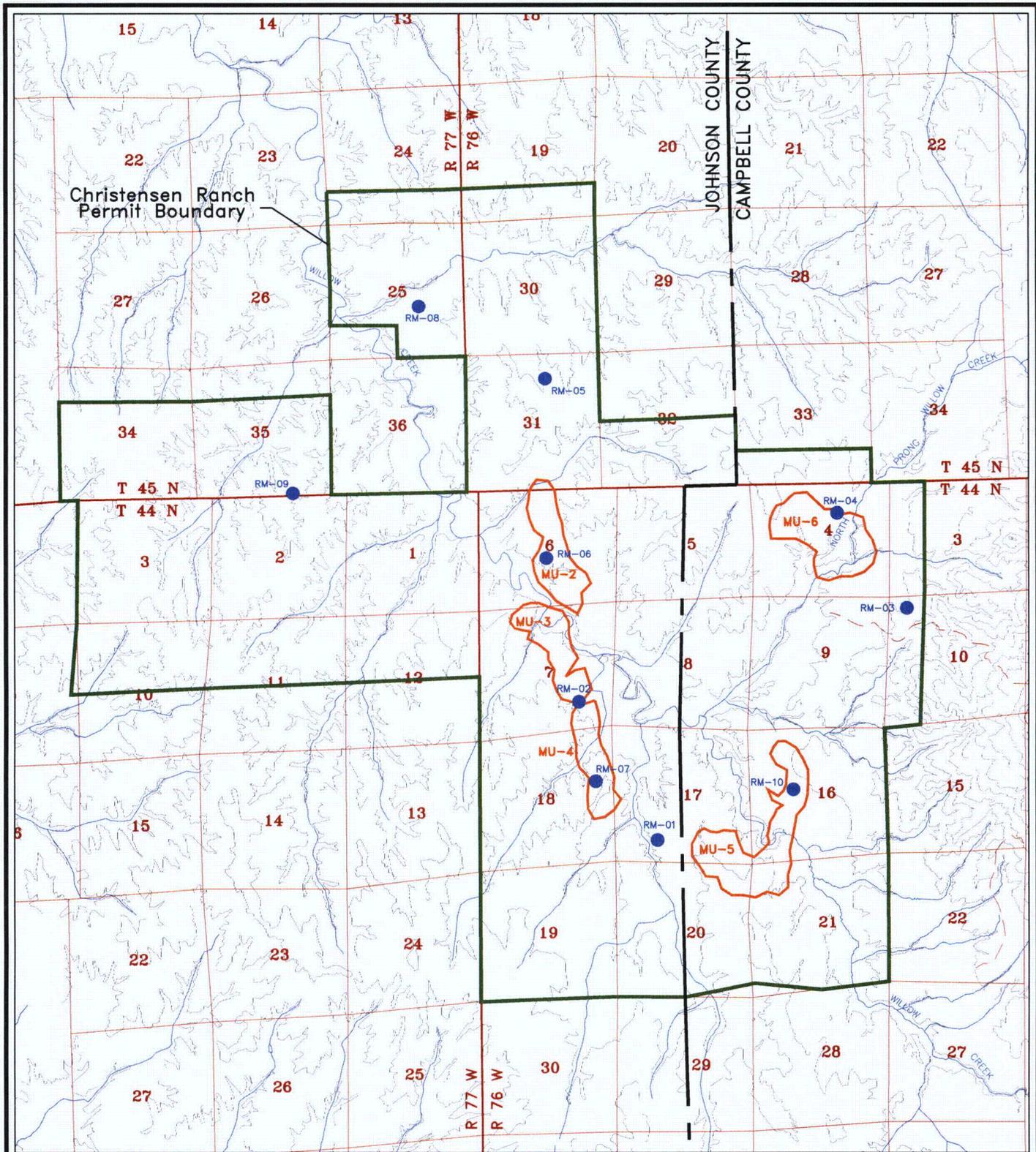
From: W.M. Sharp, 1995

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FIGURE 4-1
CHRISTENSEN RANCH PROJECT
STRATIGRAPHIC COLUMN

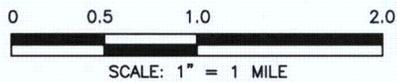
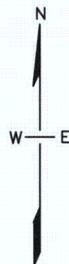
PROJECT: 217-12 DATE: APRIL 2007
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LEGEND

● RM-01 LOCATION OF REGIONAL MONITOR WELL



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FIGURE 6-1
CHRISTENSEN RANCH PROJECT
REGIONAL MONITOR WELL LOCATION MAP

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