6.3.3 Cell Cover Evolution

Research has shown that surface layers of rock on covers create a favorable habitat for deep-rooted plants in all climates, even in the desert. Depending on climatic conditions and cover design, the rock layer may act as a mulch, effectively reducing soil evaporation (increasing soil water storage) and trapping windblown dust, thereby providing the water and nutrients needed for the germination and establishment of vegetation. Vegetation is sparse but is beginning to establish on the main tailings cell cover. It consists primarily of annual weeds, but populations of perennial grasses, forbs, and deep-rooted woody plants are also establishing. An understanding of the ecology of these plant species provides clues about past and possible future changes in the condition of the disposal cell cover.

Currently, deep-rooted Siberian elm saplings and some robust fourwing saltbush shrubs grow on the cell cover, primarily on the south two-thirds of the cover (DOE controls the elm saplings with herbicide to avoid the establishment of mature trees). Their presence suggests that the underlying relocated materials and tailings are moist, particularly in that area. The sparsity of deep-rooted plants on the north portion of the cover may be because the thick layer of compacted relocated materials over the slimes is inhibiting root penetration.

The long-term consequences of changes in the ecology of covers, including the encroachment and establishment of populations of deep-rooted plant species, can be either detrimental or beneficial depending on the cover design and management practices (Link et al. 1994). A key issue is whether deep-rooted plants that establish on the cover will increase or decrease the likelihood of precipitation percolation through the cover and into the tailings. Detrimental effects are related to root growth through covers and into tailings; plants can increase percolation flux by accelerating soil development, which increases permeability by creating fissures or planes of weakness in the soil structure. Beneficial effects are related to the extraction of soil water by plants (transpiration) and erosion protection; consequently, plant encroachment could actually enhance the performance of the cover. Some studies have shown that vegetation significantly decreases percolation of precipitation (Benson et al. 2011, Waugh et al. 2009). Transpiration can greatly limit percolation if habitat characteristics favor the establishment and resilience of a diverse plant community. Therefore, a combination of high transpiration rates and erosion protection can be achieved.

Ecological succession and natural soil-forming processes alter engineered soil covers over relatively short time periods regardless of climate, cover design, or service life. Studies of disposal cell and landfill covers across the country have shown that compacted soil layers (similar to the main tailings radon barrier) typically fall short of low-permeability targets, often during or shortly after construction, and sometimes by several orders of magnitude (NRC 2011). For example, if compaction of the radon barrier achieved a permeability of 1×10^{-7} cm/s as designed, the current permeability may be closer to 1×10^{-5} cm/s.

6.3.4 Cell Cover Hydraulic Performance

The Reclamation Plan (ARCO 1990) did not reference soil physical or hydraulic property criteria for the cover. Apparently there were no criteria for the permeability of the cover or for percolation flux through the cover. However, it did provide results of grain-size analyses and *Ks* tests for samples of materials specified for use in constructing the radon barrier. Test materials were low-plasticity clay and sandy clay. Geometric means of permeability tests were 1.7×10^{-8} cm/s for the clay and 2.4×10^{-7} cm/s for sandy clay, with all samples compacted to the specified 100 percent of Standard Proctor dry density. Although the as-built permeability of the radon barrier was not measured directly, it was likely assumed that the as-built *Ks* was between 1×10^{-7} and 1×10^{-8} cm/s as noted in Section 6.3.1.

It would seem likely that percolation into tailings is potentially greatest where water ponds in depressions. However, as noted previously, evaporation appears to be the dominant factor in reducing the ponds (rather than percolation through the cover). ARCO also believed evaporation to be the dominant factor, as this was the method used to eliminate ponded fluids during the wicking procedure (Applied Hydrology Associates Inc. 1993). Even if the permeability of the radon barrier is increasing as expected, the permeability of the underlying thick layer of relocated material likely remains close to the presumed original permeability of 1×10^{-7} cm/s because those materials would not be impacted by the environmental forces affecting the surface materials.

Radon barrier permeability measurements conducted at several UMTRCA Title I sites (Waugh et al. 1999, Waugh et al. 2007, Glenn and Waugh 2001) suggest that the permeability of the radon barrier at the Bluewater site may be on the order of 1×10^{-5} cm/s. However, because of the area of the cover and expected variability of hydraulic performance of the radon barrier (due to variable thicknesses of the barrier and non-uniformity of plant growth), extensive field permeability tests would be necessary to determine the actual permeability of the radon barrier of the main tailings disposal cell. These tests are not considered to be necessary at this time because the performance criterion for radon emission is being met.

6.4 Disposal Cell Seepage

6.4.1 Conditions for Seepage

ARCO assumed that seepage of fluids remaining within the disposal cell would continue to occur after closure of the cell but did not predict the rate or quantity of seepage (ARCO 1990). The cover design was intended to control emanation of radon from the encapsulated tailings and shed precipitation runoff without causing erosion of the cover; percolation of precipitation through the cover and into the cell was not a factor in the design requirements.

Seepage from the disposal cell is controlled by a difference in total hydraulic head. Total hydraulic head is the sum of hydraulic pressure head and elevation head above a reference datum, which in this case is the bottom of the disposal cell. Because the total hydraulic head within the final disposal cell is greater than the total hydraulic head underlying the disposal cell, fluid seepage from the tailings into the underlying foundation material is expected to persist in both saturated and unsaturated conditions.

The degree of saturation within the disposal cell is a key component in evaluating unsaturated seepage. However, actual saturation within the disposal cell is unknown. Cell material saturation was not measured or estimated by ARCO and has not been measured since DOE acquired the site. For the purposes of this assessment and based on studies conducted on the Shiprock, New Mexico, Disposal Site cell (DOE 2012), it is assumed that the sand tailings are moist but unsaturated and the slimes are saturated. Although ARCO attempted to dewater the sand tailings, pumping likely did not completely drain the tailings, and precipitation would have recharged the tailings to some degree after pumping ceased in 1985.

6.4.2 Seepage Rate

The rate of seepage, or tailings fluid flow through the tailings into underlying foundation material, is governed by the hydraulic conductivity of each material composing the disposal cell. The main tailings disposal cell can be described as a layer-cake type of arrangement with the cover materials (i.e., the radon barrier and underlying soil placed to attain the final construction grade) overlying the tailings mass (sands and slimes), which in turn overlie foundation materials (alluvium, basalt, and limestone). Under saturated conditions, when the largest volume of tailings fluid flow would occur, hydraulic conductivity of the overall system is controlled by the lowest hydraulic conductivity of the materials in the layer-cake arrangement.

Table 8 provides estimates for saturated hydraulic conductivities of materials existing in and under the disposal cell that are used in this analysis. Assuming that tailings fluid seepage is controlled by natural flow, values provided in Table 8 indicate that moisture will percolate through the cover, will flow at a slower rate through the tailings, and will eventually discharge into the underlying foundation materials (which have the highest *Ks*).

Table 8. Estimated Saturated Hydraulic Conductivities for the Bluewater Disposal Cell

Layer	Vertical *Ks* (cm/s)
Cover	10^{-5}–10^{-4} [a]
Tailings: Sand[b]	10^{-6}–10^{-5}
Slimes-Sand[b]	10^{-6}–10^{-5} [c]
Slimes[b]	10^{-7}–10^{-6}
Foundation Materials: Alluvium	10^{-4}–10^{-3}
Basalt	10^{-2}–10^{-1}

cm/s = centimeters per second; *Ks* = saturated hydraulic conductivity

[a] Long-term value after soil development effects have occurred (NRC 2011)

[b] Licensing Documentation, Volume 22, Page 7 (Dames & Moore 1984b)

[c] Conservatively assumed to be controlled by interbedded sand layers

Because of the presence of sand in the slimes-sand portion of the disposal cell, that portion is conservatively assumed to have the same hydraulic conductivity as the sand portion. Therefore, the sand and slimes-sand portions are hereafter included together as "coarse tailings." The slimes portion of the cell is considered to be "fine tailings."

6.4.3 Estimated Seepage Quantity

6.4.3.1 Water-Balance Equation

For this assessment, the following basic water-balance equation is used to estimate seepage amount after construction of the cover in 1995.

$$I - O = \Delta S$$

where I = inflow
O = outflow
ΔS = change in storage

6.4.3.2 Inflow

Inflow quantity to the disposal cell is governed by site-specific climatic parameters, which are unavailable. However, average monthly precipitation quantities are available for the Grants, New Mexico, airport, and average monthly evaporation quantities are available for the region (Appendix A, Table 2). Due to the proximity of the site to the data sources, it is assumed that these quantities are representative of precipitation and evaporation at the Bluewater site. Based on these data, the Bluewater disposal cell cover receives an annual average of approximately 10.3 inches of precipitation, with an average free-surface evaporation of 45.9 inches per year.

Yearly free-surface evaporation rates are about 4.5 times greater than yearly precipitation. Evaporation exceeds precipitation in 9 months of the year. During the wettest months of July through September, evaporation rates range from 3.3 to 4.6 times greater than monthly precipitation. Not all precipitation evaporates, however; the water cycle includes infiltration into the ground (some of which recharges aquifers), uptake by vegetation, and diversion into surface water systems. Also, evaporation is an ongoing phenomenon averaged over time, whereas about half of the annual precipitation at the site occurs from July through September as high-intensity, short-duration convective storms. During such storm events, the rate of precipitation far exceeds the rate of evaporation. Regardless, evaporation in the region and at the site is a significant factor in reducing precipitated moisture.

Runoff on the disposal cell cover occurs only during rainfall events of sufficient magnitude and intensity, and after melting of significant snow accumulations. However, there is no evidence that runoff has spilled over the edge of the cell cover. Instead, cell cover runoff accumulates as ponds in depressions that have formed over the slimes area on the north portion of the cover. These ponds persist for long periods of time that correlate to the quantity of accumulated water, indicating that percolation through the cover is minimal at this location. Evaporation, therefore, is the primary cause for loss of ponded water in the depressions.

Shallow- and deep-rooted vegetation is beginning to establish on the disposal cell cover, as noted in Section 6.3.3. It can be assumed, therefore, that some percentage of precipitation percolates through the cover; apparently more over the sand tailings area than the slimes area because that is where most of the vegetation occurs. The vegetation also indicates that evapotranspiration is occurring.

There are no site-specific data to estimate the actual amount of precipitation that percolates through the cover and into the tailings. Studies performed on other covers are not directly applicable to the Bluewater site because of differences in designs, cover materials, and climate. However, percolation rates that have been measured at other sites have ranged up to 18 percent in studies by Albright et al. (2004) and up to 42 percent by Abichou el al. (1998). Rock covers such as on the Bluewater cell may act as mulch and retain moisture, which would tend to decrease evaporation and increase percolation. In the absence of site-specific data, therefore, it is reasonable to assume that between 25 and 50 percent of precipitation currently percolates through the cover. For the purposes of this assessment to estimate a range of potential seepage from the disposal cell after construction, tailings storage and outflow are calculated based on inflow quantities of 50 percent and 25 percent of precipitation.

6.4.3.3 Change in Storage

Moisture in the tailings is stored in voids within the tailings mass. A saturated volumetric moisture content is defined when all void space is occupied by moisture, and 50 percent saturation is when half the void space is occupied by moisture. Porosity is defined as the ratio of void space to the total volume of mass. Therefore, when soil is fully saturated, the volume of moisture is equal to the porosity of the soil mass.

As tailings voids drain from a saturated condition to a lower degree of saturation, negative pore pressures develop within the tailings materials. The relationship between the volumetric moisture content and negative pore pressures is provided on soil moisture characteristic curves. There are no data for moisture-holding properties of the Bluewater site tailings, nor detailed gradation data for the tailings. Therefore, tailings characteristics from the Shiprock disposal cell are used to represent Bluewater site tailings characteristics. Although mined from different geographic locations, the uranium ore processed at both sites was derived from sandstone in the Saltwash Member of the Jurassic Morrison Formation (Merritt 1971). Change in storage calculations are provided in Appendix A, Section 5.3.3.

6.4.3.4 Outflow

Assumptions and calculations to estimate outflow, or seepage, from the disposal cell are provided in Appendix A, Section 5.3.4. When moisture infiltration exceeds available storage, outflow will equal inflow. The rate of discharge will asymptotically approach the cover infiltration rate. Estimates of the volume of moisture that can potentially seep through the base of the disposal cell can be calculated using storage estimates and assumptions taken from the impoundment geometry provided in Figure 37. The approximate areas of the two types of materials covering the bottom of the disposal cell are provided in Table 9.

Table 9. Footprint of Tailings Materials

Material	Percentage of 260-Acre Footprint	Area (ft^2)
Coarse tailings	67	7,588,152
Fine tailings	33	3,737,448

ft^2 = square feet

If the tailings become saturated, drainage will occur at the influx rate if the saturated hydraulic conductivity of the tailings is greater than or equal to the influx rate. If the saturated hydraulic conductivity of the tailings is less than the influx rate, the tailings will store much of the moisture until full saturation occurs. Table 10 provides the potential annual seepage that could occur if the tailings become saturated and outflow equals inflow.

Table 10. Potential Annual Outflow if the Tailings Become Saturated

Material	**Infiltration Percentage**	**Infiltration Rate**[a]	**Area**[b]	**Outflow Volume**		**Outflow Rate**
		(ft/yr)	**(ft^2)**	**(ft^3)**	**(gallons)**	**(gpm)**
Coarse Tailings	50	0.430	7,588,152	3,262,906	24,178,134	46.0
	25	0.215	7,588,152	1,631,453	12,089,067	23.0
Fine Tailings	50	0.430	3,737,448	1,607,102	11,908,626	22.7
	25	0.215	3,737,448	803,551	5,954,313	11.3
Total	50	0.430			36,086,760	68.7
	25	0.215			18,043,380	34.3

ft/yr = feet per year; ft^2 = square feet; ft^3 = cubic feet; gpm = gallons per minute

[a] Derived from an average annual precipitation rate of 10.3 inches per year

[b] Cell cover area is approximately equal to the tailings footprint (Table 9)

Based on the results of Table 10, the extreme scenario would suggest that approximately 36 million gallons of tailings fluid could seep from the disposal cell annually if the tailings become saturated and if 50 percent of precipitation percolates into the tailings. Until the tailings become saturated, the annual seepage rate would be less and would gradually approach the saturated tailings seepage rate.

The estimated 36-million-gallon annual seepage rate is based on an assumed upper limit of precipitation percolation through the cover. As noted in Section 6.4.3.2, increasing vegetation on the cover reduces percolation of precipitation into the tailings because of evapotranspiration. As vegetation is allowed to establish (or is enhanced to accelerate establishment) on the Bluewater cell cover, the precipitation inflow could reduce to substantially less than 25 percent of annual precipitation. And, it is possible that the tailings would not become fully saturated and that seepage would remain minimal as unsaturated drainage.

6.4.4 Estimated Seepage Contaminant Mass

Constituents of concern with respect to ARCO's groundwater corrective action program, and subsequently included in the LTSP, are molybdenum, selenium, and uranium because concentrations were above background levels in monitoring wells near the disposal cell. Therefore, the evaluation of contaminant mass that seeped from the main tailings impoundment, and continues to seep from the disposal cell, addresses these constituents.

As noted previously, an estimated 5.7 billion gallons of tailings fluids seeped through the bottom of the main tailings impoundment prior to encapsulation in 1995. Contaminant concentrations in the raw tailings water, or tailings liquor, varied due to changes in milling processes and ore characteristics, but ARCO considered the concentrations listed in Table A-2 of Appendix A,

Table A-2 as representative of the tailings liquor. According to Table A-2, the liquor had a pH of 1.2 and the following concentrations for contaminants of concern: 1.33 mg/L molybdenum, 4.0 mg/L selenium, and 19.5 mg/L uranium (Dames & Moore 1981b). Contaminant concentrations in seepage from the disposal cell are expected to be less than those in the tailings liquor because of tailings dewatering activities conducted by ARCO in the 1980s and due to mixing with precipitation that is percolating through the cover and into the tailings.

The estimated mass for each constituent of concern that seeped through the bottom of the tailings impoundment prior to completion of the disposal cell cover in 1995 is provided in Table 11. Assuming the seepage volume and contaminant concentrations are representative of actual conditions, nearly 1 million pounds of uranium would have been in the fluids that seeped from the tailings impoundment.

Table 11. Estimated Seeped Contaminant Mass Prior to Disposal Cell Completion

Contaminant	Concentration	Seeped Volume		Contaminant Mass	
	(mg/L)	(billion gallons)	(billion liters)	(mg)[a]	(pounds)
Molybdenum	1.33	5.7	21.6	2.87×10^{10}	63,300
Selenium	4.0	5.7	21.6	8.64×10^{10}	190,500
Uranium	19.5	5.7	21.6	4.21×10^{11}	928,300

mg = milligrams; mg/L = milligrams per liter

[a] Concentration times seeped liters

The mass of uranium in seepage since the cover was completed cannot be estimated because seepage rates and contaminant concentrations in the tailings fluids are unknown. However, the estimated potential seepage under saturated conditions is significantly less than the seepage that occurred prior to completion of the cell cover. Likewise, although seepage may be occurring, the contribution of contaminant mass to the underlying aquifers is also significantly less than occurred prior to completion of the cell cover. The greatest contribution to groundwater contamination, therefore, occurred prior to 1995.

6.5 Disposal Cell Performance Summary

ARCO estimated that approximately 5.7 billion gallons of tailings fluid seeped through the bottom of the main tailings impoundment prior to construction of the disposal cell cover in 1995; about half of that total was projected to have occurred prior to 1960. These fluids entered the two uppermost aquifers at the site.

Tailings fluids may continue to seep from the disposal cell indefinitely. Estimated annual disposal cell seepage volumes, which are less than 1 percent of the total seepage that occurred prior to completion of the cover in 1995, assume saturated moisture conditions within the cell. However, ARCO essentially dewatered the coarse tailings between 1982 and 1985. This large volume of material would need to refill with infiltrated precipitation before saturated discharge begins. Therefore, seepage is probably occurring under unsaturated conditions and would be minimal. If vegetation is allowed to establish on the cover, evapotranspiration will reduce the volume of precipitation entering the tailings, and saturation likely would be delayed or may not occur.

The band drains that ARCO installed in the north portion of the tailings impoundment did not completely dewater the slimes—they only reduced the water content to facilitate consolidation of the slimes. Up to 15 ft of relocated sandy-clay material, similar to the material used for the radon barrier, was placed over the slimes prior to installation of the radon barrier. This thick, low-permeability layer would significantly reduce infiltration of precipitation into the slimes and keep the seepage rate low.

Depressions have formed on the disposal cell cover because the slimes portion of the tailings impoundment continued to consolidate after the cover was completed. Precipitation runoff water forms ponds in these depressions, potentially introducing a second source of fluids infiltrating through the disposal cell. Observations of the persistence of the ponds and the results of radon flux measurements, however, indicate that there has been no reduction in the performance of the radon barrier and that the ponds are reduced primarily through evaporation rather than infiltration. Therefore, the depressions and associated ponds are not indicative of additional seepage from the cell.

Considerable uncertainties are associated with the water balance and mass estimates presented in this assessment. Nevertheless, they do not negate the primary conclusion that the volume of fluid and mass of contaminants seeping from the cell since it was constructed are very small compared to the corresponding volume and mass that seeped through the bottom of the tailings impoundment prior to cell construction.

7.0 Site Hydrogeology and Water Chemistry

Though several features of the groundwater system at the Bluewater site were discussed in the assessment of regional hydrogeology in Chapter 4 and in the summary of regional water chemistry in Chapter 5, examining findings from historical and recent studies focused on the site itself can provide additional insights regarding groundwater flow and contaminant transport in the Grants-Bluewater Valley. This chapter augments the conceptual model of the study area by presenting site characterization data from investigations of the Bluewater site conducted between the late 1970s and the late 1990s. In addition to describing historical conditions that have bearing on current observations regarding uranium concentrations in the alluvial and San Andres aquifers, the earlier reports help lay the groundwork for projecting how uranium originating at the site can be expected to behave in coming years.

This chapter also includes an assessment of the water chemistry in samples DOE collected from Bluewater site wells during a recent sampling event. The water chemistry assessment helps illustrate how some of the impacts of milling processes at the site continue to influence local groundwater quality. To a large degree, the chemical impacts are similar to those previously identified by Longmire et al. (1984) as the common legacy effects of acid- and carbonate-leach milling of uranium ores in the Grants Mineral Belt. The summary of the water chemistry monitoring results is followed by an examination of chemical phenomena that influence the mobility of uranium in site groundwater. The chapter concludes with an assessment of uranium isotope data collected at both onsite wells and wells located in areas hydraulically downgradient of the site.

7.1 Local Groundwater Flow System

Geologic formations that have affected and continue to influence groundwater processes at and near the Bluewater site are the same as those identified in the regional hydrogeology chapter, including the ancestral and recent river alluvium (the alluvial aquifer), Bluewater Basalt, the Chinle Formation, San Andres Limestone and the Glorieta Sandstone (the San Andres aquifer). The Permian Yeso and Abo Formations are also present at the Bluewater site beneath the Glorieta Sandstone.

7.1.1 Aquifer Characterization

7.1.1.1 Aquifer Hydraulic Properties

Several consulting reports dealing specifically with the hydrology of the Bluewater site prior to the site being transferred to DOE contained information on hydraulic properties of geologic units underlying the site. The hydraulic properties were mostly derived from aquifer pumping tests conducted at wells tapping either the alluvial aquifer or the San Andres aquifer. Table 12 provides a summary of the results from those tests. Background information regarding several of the tests and some of the more salient test results are discussed in the following paragraphs.

Table 12. Hydraulic Properties for the Alluvial and San Andres Aquifers Derived from Aquifer Tests

Test	Pumping Well	Drawdown Data from This Well	Flow Rate (gpm)	Duration of Pumping (minutes)	Radial Distance of Observation Well (ft)	Method of Analysis or Aquifer Type	Pumping or Recovery Data	Transmissivity (ft^2/day)	Pumped Thickness (ft)	Estimated Hydraulic Conductivity[a] (ft/day)	Storativity (dimensionless)
San Andres Aquifer Tests											
SA1	Unknown[c]	USGS 5	1,740	1,500	495	Theis Solution	Pumping	414,000	150 (assumed)	2,760	0.00052
		USGS 6			2,970	Theis Solution	Pumping	307,000	150 (assumed)	2,050	0.0014
SA2	Anaconda #1	Anaconda #1	600	1,560	NA	Theis Recovery	Recovery	55,000	150 (assumed)	370	NA
		Anaconda #1	775	2,970	NA	Theis Recovery	Recovery	57,000	150 (assumed)	380	NA
SA3	Anaconda #2	Anaconda #2	1,559	1,440	0	Specific Capacity	Pumping	13,400	123	109	NA
		Anaconda #1			426	Leaky Artesian	Pumping & Recovery	26,200		213	1.3×10^{-4}
SA4	Anaconda #5	Anaconda #5	2,000	4,620	0	Specific Capacity	Pumping	8,700	173	50	NA
		Pilot Hole			33.5	Leaky Artesian	Pumping	19,900		115	3.2×10^{-4}
					33.5	Leaky Artesian	Recovery	20,200		117	6.0×10^{-5}
		Anaconda #2			135	Leaky Artesian	Pumping	66,600		385	1.2×10^{-2}
					135	Leaky Artesian	Recovery	65,200		377	1.2×10^{-2}
		Anaconda #1			514	Leaky Artesian	Pumping	54,700		316	1.3×10^{-3}
					514	Leaky Artesian	Recovery	54,700		316	8.1×10^{-4}
SA5	928	928	2,300	2,610	0	Jacob & Lohman	Pumping	50,800	61	833	NA
SA6	S(SG) GSS[b]	G(SG)	160	2816	401.5	Leaky Artesian	Recovery	3,250	124	26	5.3×10^{-5}

Table 12 (continued). Hydraulic Properties for the Alluvial and San Andres Aquifers Derived from Aquifer Tests

Test	Pumping Well	Drawdown Data from This Well	Flow Rate	Duration of Pumping	Radial Distance of Observation Well	Method of Analysis or Aquifer Type	Pumping or Recovery Data	Transmissivity	Pumped Thickness	Estimated Hydraulic Conductivity[a]	Storativity
			(gpm)	(minutes)	(ft)			(ft^2/day)	(ft)	(ft/day)	(dimensionless)
		D(SG)	——		597.5	Leaky Artesian	Pumping	3,890		31	8.9×10^{-5}
		D(SG)			597.5	Leaky Artesian	Recovery	4,530		37	2.8×10^{-4}
SA7	OBS-3	OBS-3	NA	NA	NA	NA	NA	1,700	80 (estimated)	21	NA
SA8	D(SG)	D(SG)	NA	NA	NA	NA	NA	4,200	NA	NA	1.8×10^{-4}
SA9	G(SG)	G(SG)	NA	NA	NA	NA	NA	3,260	51 (estimated)	64	5.3×10^{-5}
Alluvial Aquifer Tests											
AL1	B(M)	B(M)	82	1,440	NA	Specific Capacity	Pumping	2,000	27	74	NA
		B(M) Obs			72.5	Leaky Artesian	Recovery	3,200	27	119	1×10^{-3}
AL2	C(M)	C(M)	72	380	NA	Specific Capacity	Pumping	3,200	22	145	NA
AL3	C(M)	C(M)	NA	NA	NA	NA	NA	8195	NA	NA	NA
AL4	F(M)	F(M)	NA	NA	NA	NA	NA	1780	8	223	NA
AL5	T(M)	T(M)	NA	NA	NA	NA	NA	5310	19	287	NA

ft = feet; ft/day = feet per day; ft^2/day = square feet per day; gpm = gallons per minute; NA = not available or not applicable

[a] Hydraulic Conductivity = Transmissivity/Pumped Thickness

[b] GSS—Aquifer test results represent the Glorieta Sandstone

[c] Gordon (1961) reported that the pumping well for test SA1 was located in the southeast corner of Section 30, T12N, R10W; location given was near the current location of well B00518.

Hydro-Search (1977) summarized the results of aquifer tests performed in earlier years at near-site wells screened in the San Andres aquifer, as previously reported in Gordon (1961). A 25-hour test in 1950 using a 1,745 gpm discharge rate at well B00518, about 2 mi south-southeast of the main tailings impoundment (Figure 18, Plate 7), resulted in estimated transmissivities that ranged from 267,00 to 455,000 ft^2/day. Two of the computed transmissivities are listed under Test SA1 in Table 12. Computed storativities using the test data varied from 5.2×10^{-4} to 1.4×10^{-3}, which are within the expected range for confined conditions. These results suggest that the San Andres aquifer south of the site and closer to the Zuni Mountains is quite productive, most likely due to the presence of fractures and solution features in limestone.

It was surmised at the time of the SA1 test pumping that well penetration was about one-third to one-half of the local total aquifer thickness. Assuming that the aquifer thickness was 300 ft, division of the transmissivities listed in Table 12 by half that length results in estimated hydraulic conductivities of 2,760 and 2,050 ft/day, values that comport with the magnitude of hydraulic conductivities associated with karst limestone in Section 4.2.4.

The production well Anaconda #1, screened in the San Andres aquifer south of the main tailings impoundment (Figure 17, Plate 7), was tested in 1951 at two different pumping rates without the benefit of observation wells (test SA2 in Table 12). Using analyses of water-level recovery data after pumping stopped resulted in derived transmissivities for the well that exceeded 50,000 ft^2/day. The pumping well penetrated about 150 vertical feet of the estimated 270 to 280 ft total thickness of the San Andres aquifer in the tested area. Division of the two transmissivities listed in Table 12 by this length produced estimated hydraulic conductivities of 370 and 380 ft/day.

Hydro-Search (1977) conducted test SA3 (Table 12) as part of its initial hydrologic investigation of the Bluewater site. The test consisted of pumping well Anaconda #2 for 24 hours at a constant rate of 1,559 gpm. Using drawdown data from both the pumping well and an observation well (Anaconda #1) about 425 ft south of the pumping well resulted in derived transmissivities of 13,400 and 26,200 ft^2/day, the latter of which Hydro-Search (1977) assumed to be most representative of the San Andres aquifer at the Bluewater site. The aquifer storativity derived from analysis of the test data was 1.34×10^{-4}, a value that was again representative of a confined aquifer. Hydro-Search (1977) concluded that the transmissivity and storativity values resulting from the test were generally representative of the San Andres aquifer in the vicinity of the Bluewater site. Estimated hydraulic conductivities from the test are 109 and 213 ft/day (Table 12).

Two additional tests conducted at wells screened in the San Andres aquifer with pumping durations of more than a day (test SA4 and SA5) resulted in estimated hydraulic properties for the aquifer that were similar in magnitude to those mentioned above for the aquifer. Test SA4 was based on pumping well Anaconda #5 for more than 3 days and made use of drawdown data collected at 3 observation wells (Dames & Moore 1981c). Estimated hydraulic conductivities for the aquifer using seven different analyses of the test data varied from 50 to 385 ft/day (Table 12). Test SA5 is of particular interest in this study because it was conducted a few miles east and downgradient of the east border of the Bluewater site, along a flow path extending east-southeast from site property directly north of the main tailings disposal cell (Figure 18). The test pumping, conducted at well 928, about 0.6 mi north of the large tailings disposal cell at the Homestake site, resulted in a derived transmissivity of 50,800 ft^2/day and an estimated hydraulic conductivity of 833 ft/day. These latter results suggest that karst features are present in the San Andres aquifer in the vicinity of the Homestake site as well as at the Bluewater site and support the possibility of relatively short travel times (<20 years) between the two sites (see Section 4.3.3.2).

Aquifer hydraulic properties derived from aquifer test SA6 (Figure 17, Plate 7), which used well S(SG) east of the main tailings disposal cell as the pumping well, are of special interest to this study because they are considered representative of the Glorieta Sandstone specifically (Applied Hydrology Associates Inc. 1990). Barrier boundary effects were observed during this test, presumably due to the presence of formation offsets at the Ambrosia Lake and East-West Faults. The data were analyzed by Dames & Moore (1981c), resulting in estimated transmissivities from 3,200 to 4,500 ft^2/day and storativities varying from 5.3×10^{-5} to 2.8×10^{-4} (Table 12). In addition to producing transmissivities for the sandstone that were lower than those considered representative of shallower, fractured limestone, assessment of the test data indicated that fully unconfined conditions were not locally present, at least not in the Glorieta Sandstone.

Applied Hydrology Associates Inc. (1990) also estimated aquifer parameters for the San Andres aquifer using data from short-duration, single-well hydraulic tests performed by ARCO in well OBS-3, and two additional, now-abandoned wells D(SG) and G(SG) just east of the main tailings impoundment (tests SA7, SA8, SA9). The transmissivities resulting from the test analyses (1,700 to 4,200 ft^2/day) were lower than most of the transmissivities derived from long-duration tests (e.g., Tests SA1 through SA5) in the San Andres aquifer. It is possible that the relatively low transmissivities stemming from the singe-well tests were also representative of the lower hydraulic conductivities expected in the Glorieta Sandstone in comparison to those reflective of karst limestone.

All of the hydraulic conductivities listed in Table 12 stemming from aquifer testing in the San Andres aquifer fall into the range of conductivities reported by Huntoon (1995) for the results of numerous pumping tests in karst aquifers. In addition to giving credence to the transmissivity values presented in the table, this result lends support to the travel time calculations discussed in Section 4.3.3.2 regarding the San Andres aquifer.

Hydro-Search (1977, 1978) posited that a wide range of hydraulic properties derived for the San Andres aquifer from multiple aquifer tests near the Bluewater site was an indicator of significant aquifer heterogeneity. Transmissivities were expected to vary spatially depending on a number of factors, including (1) the degree of fracturing or limestone dissolution in the aquifer, (2) the continuity of solution openings in limestone, and (3) the proportion of sandstone within the vertical domain occupied by the San Andres Limestone and Glorieta Sandstone. In general, transmissivity was expected to increase as the proportional amount of sandstone decreased.

To develop estimates of the hydraulic properties of the alluvial aquifer at the site, Dames & Moore (1981c) analyzed drawdown data from a day of pumping at B(M) (test AL1), a well just southwest of the main tailings impoundment (Figure 17 and Plate 7) and that is now abandoned. The collected data showed evidence of leaky-confined conditions, and the computed storativity was 1×10^{-3} (Table 12), which is smaller than expected if the alluvial aquifer was releasing groundwater under unconfined conditions. A hydraulic conductivity of about 120 ft/day was derived for the aquifer, and additional analyses of the data resulted in an estimated vertical hydraulic conductivity for the overlying Bluewater Basalt of about 13 ft/day. This latter value suggested that fractures in the basalt rendered it moderately permeable and capable of conveying significant quantities of downward seepage due to infiltration of water in shallow surface alluvium above the basalt.

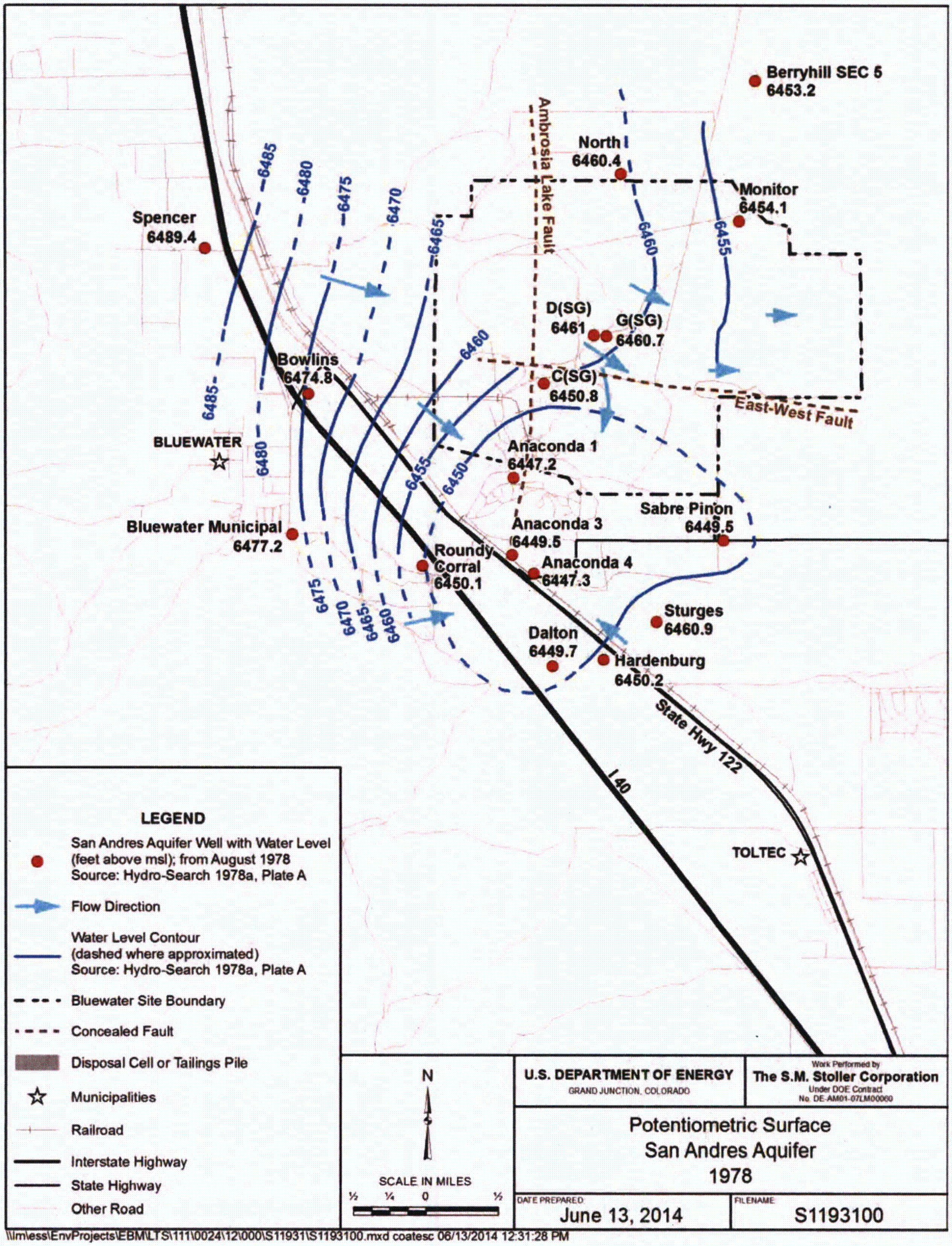
A second aquifer test conducted in the alluvial aquifer and analyzed by Dames & Moore (1981c) was based on pumping from alluvial well C(M) (Test AL2), which was about 500 ft south of the main tailings impoundment (Figure 17 and Plate 7) but is now abandoned. The well was pumped at a constant rate of 72 gpm, but groundwater withdrawal was stopped after a relatively short duration of 6 hours because the well could no longer sustain the pumping rate. An estimated transmissivity of 3,200 ft^2/day was derived from drawdown data, suggesting that well C(M) was screened in coarse-grained, highly permeable alluvial materials deposited in an ancestral Rio San Jose paleochannel. Both barrier and recharge boundary conditions were observed in the drawdown curves generated by the testing. The barrier effects were likely created by the pumping cone-of-depression contacting the north boundary of the ancestral river alluvium north of well C(M). It was impossible to tell whether unconfined flow conditions were present during the testing because no observation well data were available for calculating aquifer storage parameters.

Transmissivities derived by Applied Hydrology Associates Inc. (1990) from individual well, short-duration tests at alluvial aquifer wells F(M), T(M), and the now-abandoned C(M) (tests AL3, AL4, AL5) ranged from about 1,800 to 8,200 ft^2/day. These latter values were of the same general magnitude as those mentioned above for earlier tests at alluvial wells B(M) and C(M), and were considered indicative of highly permeable sands and gravels in paleochannels of the ancestral Rio San Jose.

In general, the transmissivity, hydraulic conductivity, and storativity values derived from testing of the alluvial and San Andres aquifers in the vicinity of the Bluewater site comported with values for these parameters discussed in Chapter 4.

7.1.1.2 Effects of Pumping Anaconda Production Wells

The high pumping rates from the San Andres aquifer achieved during the aquifer tests at wells Anaconda #1, Anaconda #2, and Anaconda #5 (Tests SA2, SA3, and SA4 in Table 12) suggest that all of the Anaconda production wells were pumped at relatively high rates (600 to 2,000 gpm) during milling years at the Bluewater site. As mentioned in Section 4.3.3, the high rates of groundwater removal produced a wide (>1 mi) cone of influence that appeared to be centered on the Anaconda production wells. The general dimensions of the cone of influence during milling years are shown in Figure 42, which presents a map view of the potentiometric surface in the San Andres aquifer based on water-level measurements in August 1978 at wells screened in the aquifer in the vicinity of the Bluewater site. As indicated in the figure, the drawdowns created by the pumping in August 1978, in comparison to the ambient potentiometric surface, appeared to be about 2.5 to 3 ft. Though the actual pumping rates at the Anaconda production wells at the time were unknown, this relatively low amount of drawdown, in comparison to the total aquifer thickness south of the East-West Fault, suggested that the aquifer is very prolific in areas that were tapped by the Anaconda wells.



Source: Hydro-Search 1978

Figure 42. Potentiometric Surface in the San Andres Aquifer in August 1978

Hydro-Search (1978) developed the flow arrows shown in Figure 42 on the basis of the observed water levels in local wells and a general understanding of how the groundwater system responded to pumping from the San Andres aquifer at and near the Bluewater site. In addition to showing convergent flow toward the area containing the Anaconda production wells, the arrows indicated southward groundwater movement across the East-West Fault, providing a mechanism for contaminant migration from beneath the main tailings impoundment to areas several hundred feet south of the East-West Fault. The detection of uranium at wells Anaconda #1 and Anaconda #5 at concentrations on the order of 0.2 to 0.35 mg/L in 1980–1981 (Hydro-Search 1981a) indicated that the southward migration of uranium contamination across the fault, via downward conduit flow in the fault zone, was possible. Similarly, the detection of uranium at production well Anaconda #1 in August 2013 at a concentration of about 0.09 mg/L suggested that the contamination that had migrated to the area containing this well as early as 1980 was continuing to reside in local portions of the San Andres aquifer some 33 years later. This latter uranium detection combined with a measured uranium concentration of about 0.13 mg/L at well 18(SG) (Figure 16) in 2013 suggested a possibility that uranium contamination originating at the main tailings impoundment had, in the past, migrated at least 0.8 mi south of the East-West Fault and persists in this general area today.

Inspection of geologic cross section A-A' in Figure 10 indicates that the vertical offset at the East-West Fault just east of the main tailings disposal cell is about 370 ft. The resulting juxtaposition of the San Andres aquifer north of the fault with the mostly impermeable Chinle Formation on the south side suggests that the fault provides a significant partial barrier to southward contaminant migration to areas containing production wells Anaconda #1 and Anaconda #5 (see Figure 17, Figure 42). Nevertheless, southward groundwater flow and concomitant contaminant migration across the East-West Fault appears to have occurred in the past, presumably due to the increased hydraulic gradient across the fault induced by pumping at the Anaconda production wells. Because hydraulic heads in the San Andres aquifer north of the East-West Fault continue to be about 5 to 10 ft higher than comparable heads just south of the fault (Section 4.3.3, Figure 23), long after pumping from the Anaconda production wells ceased, continued southward migration of contamination across the fault appears possible.

Further inspection of geologic cross section A-A' in Figure 10 suggests that, for southward groundwater flow across the East-West Fault in the San Andres aquifer to occur, downward, conduit flow within the fault zone is required. This finding comports with observations by Bense and Person (2006) that faults frequently influence groundwater flow by acting as both partial barriers and conduits. Downward groundwater flow and contaminant migration in the East-West Fault on the west side of the Ambrosia Lake Fault also appears possible. This can be seen in geologic cross section D-D' in Figure 11, which shows the fault providing a connection between ancestral river alluvium and the San Andres Limestone south of the fault. In this latter case, conduit flow within the fault zone would have provided a pathway for contamination that historically originated as downward percolation of tailings fluids into the Bluewater Basalt, to subsequently migrate to alluvium and then deeper to the San Andres aquifer.

It is possible that the relatively high uranium concentrations at wells Anaconda #1 and Anaconda #5 in 1980–1981 were mostly caused by southward conduit flow along the Ambrosia Lake Fault, as opposed to flow across the East-West Fault via downward conduit flow in the East-West Fault zone. If this were the case, the presence of a relatively high uranium concentration in 2013 about 2,000 ft east of the southern end of the Ambrosia Lake Fault, at well 18(SG) (Figure 16), would have been caused by gradual eastward migration of uranium from the Ambrosia Lake Fault. Further investigation of flow phenomena in the vicinity of the intersection of the two faults and areas south of the East-West Fault may someday reveal the actual processes involved.

7.1.1.3 Aquifer Flow Conditions

Applied Hydrology Associates Inc. (1990) surmised that virtually all groundwater flow in the San Andres aquifer at the site occurs under dual-domain, or fracture-matrix, conditions (e.g., Zimmerman et al. 1993) associated with karst features in the San Andres Limestone. This position assumes that all horizontal groundwater movement takes place in preferential pathways consisting of solution channels and fractures in limestone, and that all remaining groundwater in the adjacent rock matrix, consisting of limestone and sandstone, is effectively immobile. In such a flow system, exchange of dissolved contamination between the fractures and the matrix takes place either through molecular diffusion in the matrix or very slow advection (movement with moving water). Though such dual-domain processes undoubtedly occur in the San Andres Limestone portion of the aquifer, it is unlikely that they govern all significant transport processes in the aquifer. Given that the San Andres Limestone and Glorieta Sandstone have a combined thickness of about 250 to 350 ft in the vicinity of the site, and that fractured sandstones in the aquifer are sufficiently permeable to facilitate relatively rapid advection of contaminants in a horizontal direction, the conceptual model in this study assumes that San Andres aquifer groundwater migrates horizontally in fractured sandstone (Section 4.3.3.2) as well as in karst features in limestone. Additionally, the groundwater migrates horizontally at variable rates that both increase and decrease with depth in the aquifer. Average linear velocity calculations in Table 5 suggest that the advective travel times in fractured sandstone between the Bluewater site and areas hydraulically downgradient of the site are only moderately longer than those attributed to advective transport in karst limestone features (e.g., solution channels). A wide range of estimated groundwater velocities in these two types of media suggests that the arrival times at downgradient sites for contaminants migrating eastward and southeastward in the San Andres aquifer could span decades if not hundreds of years.

While investigating contaminant plumes at the Bluewater site, Hydro-Search (1977, 1978) concluded that the groundwater velocities controlling contaminant migration in both the alluvial and San Andres aquifers were on the order of hundreds of feet per year. In comparison, Dames & Moore (1986a, 1986b) concluded that groundwater velocities in the two aquifers were greater than 1,000 ft/yr and could be higher than 5,000 ft/yr in some locations. In addition, Dames & Moore deduced that relatively stable concentrations of constituents observed at locations on the Bluewater site were caused mostly, if not entirely, by transverse dispersion. This finding suggested that the rate at which contamination was being released to the plumes was relatively constant, and that the leading portions of contaminant plumes had migrated rapidly offsite. Both Hydro-Search (1977, 1978) and Dames & Moore (1986a, 1986b) identified transverse dispersion as the primary mechanism for attenuating onsite concentrations with transport distance in the alluvial and San Andres aquifers. Applied Hydrology Associates Inc. (1990) attributed contaminant attenuation on and downgradient of the Bluewater site to several possible processes, including transverse dispersion.

Assuming that average linear velocities in both aquifers were high, Dames & Moore (1986a) calculated that the leading portion of Bluewater site–related contaminant plumes would migrate offsite within periods of less than 10 years after initial contaminant mobilization. This conclusion was supported by their transport modeling of the relatively conservative (nonreactive) anions chloride and sulfate, as well as TDS. The high velocities simulated in the Dames & Moore models were achieved by assuming that the effective porosity of the San Andres aquifer was 0.02, a value that is much smaller than the porosities of true porous media. Without using such a low effective porosity, Dames & Moore found that the transport models were incapable of matching elevated concentrations of chloride, sulfate, and TDS observed at San Andres aquifer wells along the east border of the Bluewater site in the 1980s.

Hydro-Search (1977, 1978) and Dames & Moore (1986a) concluded that downward seepage of tailings fluid from the base of the main tailings impoundment provided the primary source of contamination that was transported east and southeast from the Bluewater site in the alluvial and San Andres aquifers. Downward seepage of tailings fluids west of the Ambrosia Lake Fault through Bluewater Basalt and subsequently to underlying ancestral Rio San Jose alluvium provided the source of contamination transported within the alluvial aquifer. Because hydraulic heads in the alluvial aquifer were generally about 20 to 40 ft higher than equivalent heads in the San Andres aquifer east of the tailings impoundment, it was also likely that some alluvial contamination migrated eastward across the Ambrosia Lake Fault and north-northeastward across the East-West Fault into the San Andres aquifer in the vicinity of San Andres hill (where the San Andres Limestone/Glorieta Sandstone sequence is in contact with the alluvium because of the uplifted fault block). In addition, downward seepage of tailings fluids directly to the San Andres aquifer appeared to occur where the base of the tailings in the southeast corner of the main tailings impoundment was either in direct contact with the underlying San Andres Limestone or separated from the San Andres Limestone by a thin layer of jointed (porous) basalt.

Downward flow of tailings fluids to the alluvial and San Andres aquifers is estimated to have declined significantly since milling operations ceased (Chapter 6). Nonetheless, mechanisms for steadily feeding contamination to plumes migrating from the Bluewater site in the alluvial and San Andres aquifers are apparently still present (Appendix A).

7.2 Groundwater Chemistry in the Vicinity of the Bluewater Site

7.2.1 General Water Chemistry

The water chemistry at wells on the Bluewater site was examined partly to identify how the presence of onsite contaminant sources affects water quality. This was largely accomplished by comparing the chemistry of onsite groundwater samples with the samples used to evaluate regional water chemistry in Chapter 4. All onsite wells (background, contaminated, uncontaminated) were included in the onsite groundwater geochemistry evaluation. Samples used in this analysis were collected from May 14 through May 16, 2013, at the locations shown in Figure 43 (wells T(M) and X(M) were dry).

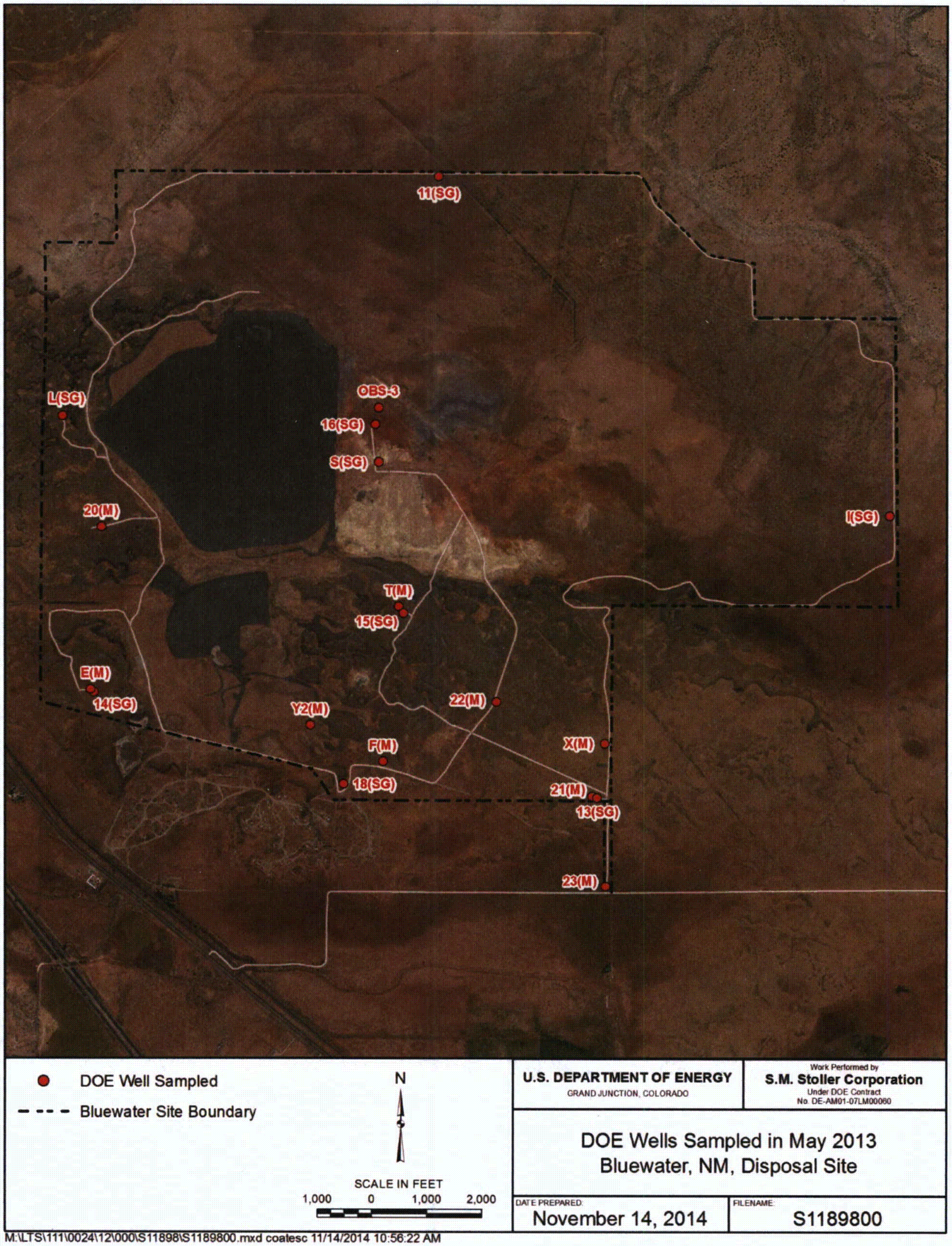


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Figure 43. Locations of Bluewater Site Wells Sampled in May 2013

7.2.1.1 Alluvial Aquifer

Oxidation states measured in the groundwater samples collected from alluvial aquifer wells were relatively high, with all but one of the pE values falling above 4.6. The calculated pE value at well E(M) was 0.7. DO concentrations in alluvial groundwater samples ranged from 1.6 to 6.5 mg/L, except in the sample from well E(M), which had a DO concentration of 0.1 mg/L. Thus, the combination of DO concentrations and pE values indicate that groundwater in the vicinity of well E(M) has a lower oxidation state than the groundwater monitored at other alluvial wells.

Water samples collected from all but one of the alluvial aquifer wells showed nearly neutral pH values, ranging from 7.02 to 7.40; the pH at well E(M) was 8.05. The salinity of alluvial aquifer groundwater was moderate, with measured specific conductivity values ranging from 547 to 1,859 µS/cm.

A Piper diagram for the alluvial well samples, shown in Figure 44a, shows that cation equivalents in the cation triangle form a linear trend along a line extending from the sodium (plus potassium) corner to the calcium-magnesium border. The calcium-to-magnesium ratio is about 4 to 1. Most of the anion compositions are about midway between sulfate and bicarbonate, with one sample containing nearly all sulfate. Chloride is low in all onsite alluvial aquifer samples (Figure 44a). Both cation and anion compositions are similar to those for alluvial aquifer groundwater samples collected at most offsite wells in the region. Exceptions to this general finding occur at regional wells in San Mateo Creek alluvium north of the Homestake site, which are noticeably higher in salinity and sulfate than any onsite alluvial aquifer samples. This is because the alluvial aquifer north of the Homestake site has been impacted by contaminated water from the Ambrosia Lake Valley area. The mining and milling activities that generated contamination in the Ambrosia Lake Valley were separate from the uranium milling that took place at the Bluewater site.

The alluvial aquifer groundwater samples group tightly on a plot of calcite versus gypsum saturation indexes, with the exception of the sample from E(M) (Figure 45). Most of the samples are slightly undersaturated with respect to calcite but are undersaturated by about 1 to 2 orders of magnitude with respect to gypsum. In contrast, the sample from well E(M) is far undersaturated with calcite and close to saturation with gypsum. These results suggest that most of the alluvium contains calcite, probably in the form of limestone grains or carbonate cements, whereas the alluvium is devoid of gypsum.

The uranium concentrations in two samples of alluvial groundwater, from wells 21(M) and 22(M), were relatively high at 148 and 380 micrograms per liter (µg/L), respectively. These concentrations comport with monitoring results from previous sampling events, as both wells have shown site-related uranium contamination since their installation in summer 2011. Water in the samples from the two wells also had the highest relative sodium equivalents, suggesting that site-related groundwater contamination elevates sodium concentrations in comparison to inflowing groundwater from offsite areas.

Molybdenum concentrations in all alluvial aquifer samples were low, ranging from 0.3 to 8.4 µg/L. Nitrate (as NO_3) concentrations at all but two wells were also relatively low, ranging from less than 0.1 to 15.3 mg/L. Nitrate concentrations in the water samples from 21(M) and 22(M) were 43.4 and 161.2 mg/L, respectively, indicating that alluvial aquifer groundwater that has been impacted by contamination contains elevated levels of this constituent as well.

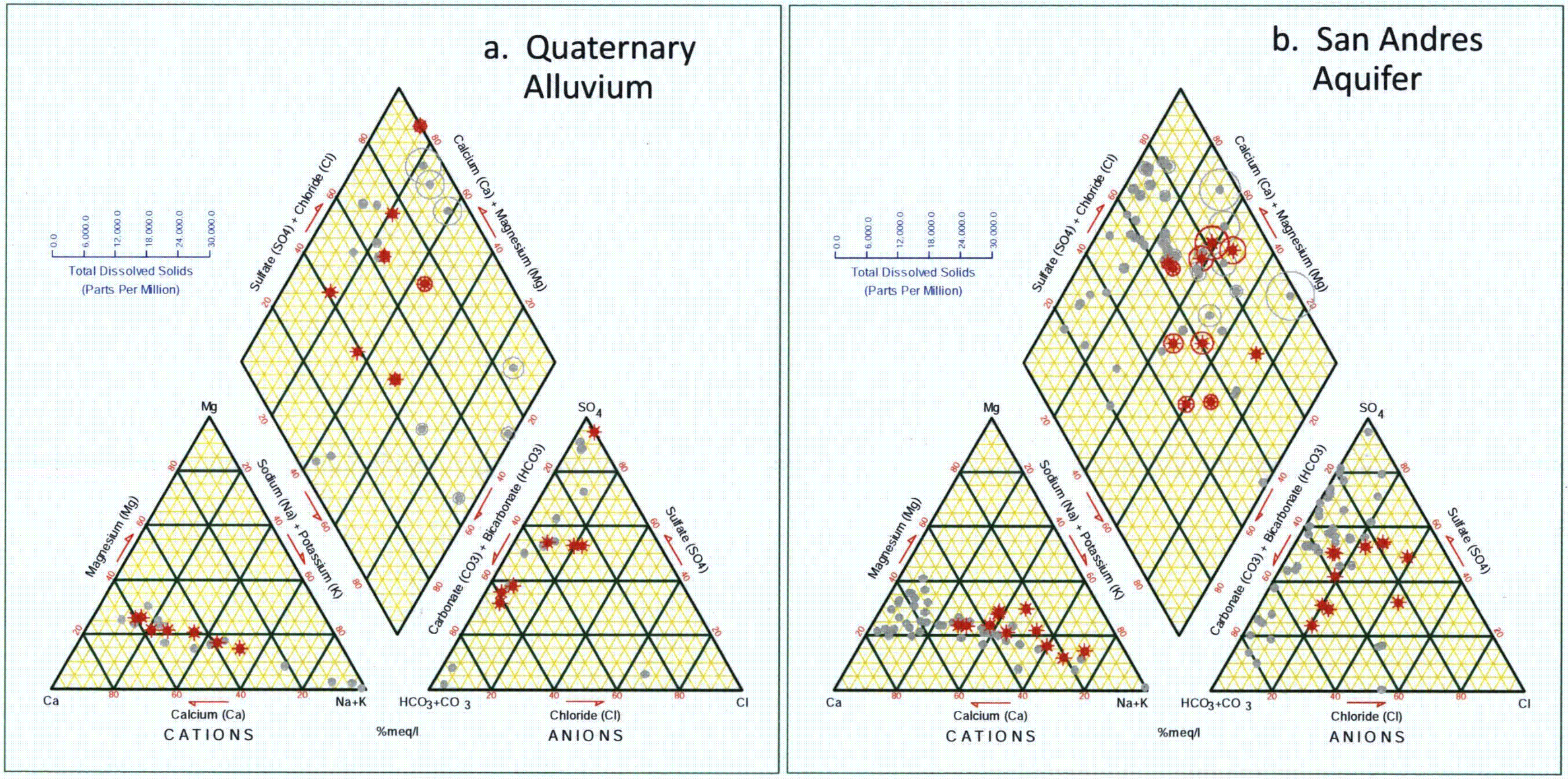

Figure 44. Piper Diagrams for Onsite Groundwater Samples: (a) Alluvial Aquifer and (b) San Andres Aquifer (Gray symbols represent the regional background chemistry from samples collected at offsite well locations, and red symbols represent samples collected from onsite wells.)

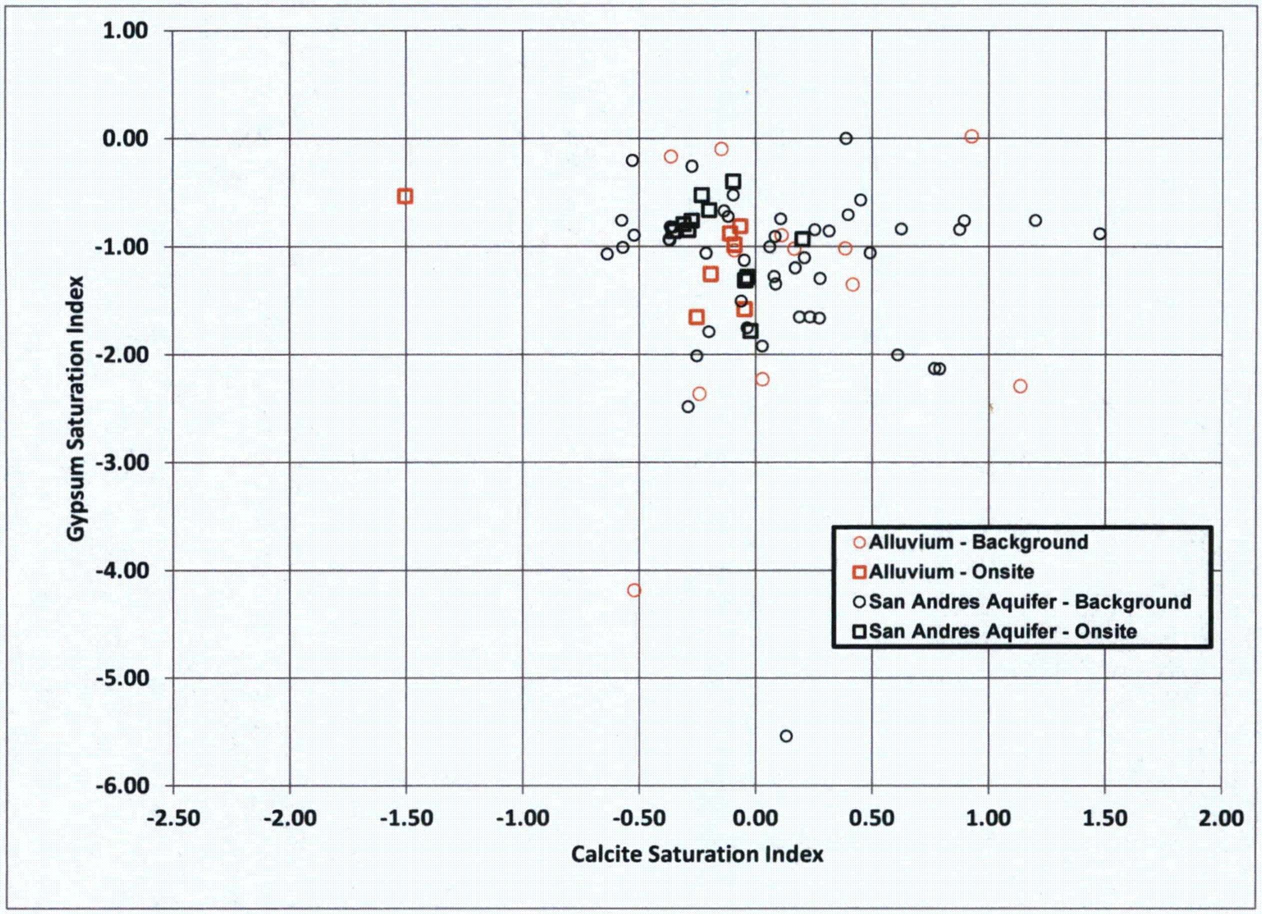

Figure 45. Saturation Indexes for Calcite and Gypsum in Onsite Alluvial Aquifer and San Andres Aquifer Samples

7.2.1.2 San Andres Aquifer

Analytical results from the May 2013 sampling of 10 San Andres aquifer wells on the Bluewater site indicated that the aquifer has moderate to relatively high salinity. Values of specific conductivity varied from 1,204 to 4,065 µS/cm, with the lowest value observed at a depth of about 235 ft below the top of casing in well I(SG), on the east border of the site (Figure 43). In comparison, specific conductivities in regional, background groundwater hydraulically upgradient of the Bluewater site range from less than 1,000 to 2,000 µS/cm (Table 6, Section 5.2.4).

With the exception of well I(SG), the pH values at the San Andres aquifer wells were near neutral, ranging from 6.53 to 7.01. A pH of 7.97 was measured at the 235 ft depth in this well.

The above-mentioned, unique chemical signatures of salinity and pH at well I(SG) reflect the fact that chemical stratification has been identified in this well. Specific conductivities have been shown to vary widely in the well depending on the depth below ground surface at which samples are collected (Section 10.2.2). Uranium concentrations measured in three samples collected at different depths in well I(SG) in May 2013 also show evidence of chemical stratification. The shallowest sample, collected near the top of the water column in the well at a depth of about 235 ft below the top of casing, had a uranium concentration of about 0.005 mg/L. Another sample, collected at a depth of about 255 ft below the top of casing, had a uranium concentration of 0.350 mg/L. The third sample, collected at a depth of about 300 ft below top of casing, had a uranium concentration of 0.344 mg/L. These results suggested that it is very important to collect multiple samples at different depths from well I(SG) if the data are being used to assess the rate at which site-related contamination might be attenuating.

Uranium concentrations at the onsite San Andres aquifer wells ranged from 0.003 to 1.450 mg/L. Water collected from six wells (13(SG), 16(SG), 18(SG), I(SG), OBS-3, and S(SG)) had uranium concentrations greater than 0.1 mg/L, which are indicative of site-related contamination. Molybdenum concentrations were low in all 10 San Andres wells, ranging from less than 0.2 to 3.8 µg/L. Nitrate concentrations were also relatively low, ranging from less than 0.08 to 20.3 mg/L (as NO_3).

In a Piper diagram of anion and cation concentrations at the onsite San Andres wells (Figure 44b), cations were shown to vary from near the sodium (plus potassium) corner to about 20 percent sodium. Sodium equivalents in the onsite samples were higher than comparable values in the regional offsite wells (Figure 33d). The calcium-to-magnesium ratio in the San Andres aquifer samples was about 4 to 1. The anion distribution in the onsite San Andres aquifer wells (Figure 44b) indicated that chloride was present in higher proportions than observed in regional background wells, suggesting that a sodium-chloride component was added to onsite groundwater in the San Andres aquifer. The logical source for this apparent addition was the former tailings impoundment at the Bluewater site.

As mentioned in Chapter 5, fine-grained pyrite is present in the sandstones within the San Andres aquifer. As a reduced sulfide mineral, pyrite has the capacity to lower the oxidation state of groundwater in the bedrock aquifer. It is possible that pyrite in the aquifer is limited to the top of the San Andres Limestone (see Section 3.3.2.4). If so, groundwater in wells completed in the middle and lower portions of the aquifer is likely to be oxidized. The pE values in onsite groundwater samples collected in May 2013 varied from 2.0 to 5.5, indicating relatively oxidized conditions. DO concentrations in the May 2013 samples ranged from 0.35 to 6.57 mg/L, values that are consistent with oxidizing conditions.

Calcite was undersaturated in several San Andres aquifer samples, but most samples were close to saturation (Figure 45). These results indicate that onsite groundwater in the aquifer is nearly saturated with calcite, which is likely caused by contact with limestone and with calcite cementation in the sandstones. The May 2013 samples were are also slightly undersaturated with respect to gypsum, but were closer to saturation than indicated in the regional background waters of the San Andres aquifer (Figure 45).

7.2.2 Uranium Mobility in Site Groundwater

Many chemical and biochemical processes affect the transport of uranium by groundwater. Uranium is immobile and not available to enter groundwater when it is attached firmly to mineral phases in the aquifer. While in the aqueous phase, uranium is mobile and can be transported long distances. Uranium can also migrate into less mobile water contained in low-permeability sediments and micropores that are separated from the preferential flow paths in an aquifer. This uranium can "bleed" back out into the groundwater and cause later contamination through a process commonly referred to as back-diffusion.

Uranium can be transferred from the aqueous phase to solid mineral phases via adsorption or mineral precipitation. Transfer to the aqueous phase is usually favored by high pH values with abundant dissolved inorganic carbon or at low pH values without dissolved carbon. Uranium forms soluble complexes with carbonate ions. Under oxidized conditions, it is least soluble at near-neutral pH. Aqueous uranium complexes can also form from dissolved organic species, although the effect of uranium complexation in most natural groundwater is less than the effects of dissolved inorganic carbon.

Uranium is sensitive to changes in the oxidation-reduction (redox) state in an aquifer because it can form oxidized U(VI) species, referred to as uranyl, and reduced U(IV) species, referred to as uranous. A special case of mineral precipitation is the formation of low-solubility uranous minerals if the oxidation state is lowered sufficiently. Uranium can also be incorporated as a trace component in alkaline-earth-bearing minerals, such as calcite that may precipitate in the aquifer.

7.2.2.1 Uranium Adsorption

Adsorption of uranium occurs at mineral surfaces. Adsorption is likely to be a dominant mechanism for uranium retardation in oxidized aquifers. Minerals with high surface area are more adsorbent than those with lower surface area. Thus, clay minerals and hydrous ferric oxide ($Fe(OH)_3$) typically bind more uranium than detrital silicate minerals such as quartz or feldspar. These fine-grained minerals often compose the matrix material that binds detrital grains together. They also commonly occur as grain coatings. Sediments sometimes contain organic particles that can also adsorb uranium.

Many studies of uranium adsorption to mineral surfaces have been conducted. In particular, adsorption to $Fe(OH)_3$ is well understood. Most research indicates that uranium forms chemical complexes with negatively charged hydroxide sites at the surface of $Fe(OH)_3$. The ability to attach to the mineral surface is largely a function of the pH and the concentration of dissolved inorganic carbon. Hydrous ferric oxide is one of the most adsorptive minerals for uranyl in nature. In rocks with little hydrous ferric oxide, other minerals such as clays may dominate uranium adsorption. Although other minerals are less adsorptive than hydrous ferric oxide, they tend to respond to pH and aqueous carbonate concentrations in patterns similar to those observed for $Fe(OH)_3$.

Geochemical modeling was used to help evaluate the potential for adsorption processes to affect the mobility of uranium. The modeling was conducted with the geochemical computer code PHREEQC (Parkhurst and Appelo 2013). The WATEQ4F thermodynamic database supplied with the PHREEQC code was used, with uranyl surface complexation constants from Mahoney et al. (2009), and calcium-magnesium uranyl carbonate complexes from Dong and

Brooks (2006). This database was selected because Mahoney et al. (2009) found that it provided the best fit to all available uranyl surface complexation data for $Fe(OH)_3$. As a check, Mahoney et al. (2009) substituted the aqueous uranyl species stability constants published by the Nuclear Energy Agency (Guillaumont et al. 2003) and found that they produced similar results.

Figure 46 shows aqueous uranium concentrations in micrograms per liter in the presence of a system containing 1 gram (g) of $Fe(OH)_3$ for each liter of water and 2,380 µg of uranium per liter of water. As shown in this figure, the uranium concentrations are also a function of the aqueous dissolved inorganic carbon concentration, which is reflective of the amount of dissolved calcite ($CaCO_3$) in groundwater. When all of the uranium mass is dissolved in the water, the aqueous-phase concentration of the uranium is 1 millimol per liter. The concentration of $Fe(OH)_3$ used in this exercise is equivalent to about 100 parts per million (ppm) of iron in the rock. For comparison, the average crustal abundance of iron is 50,000 ppm (Mason and Moore 1982).

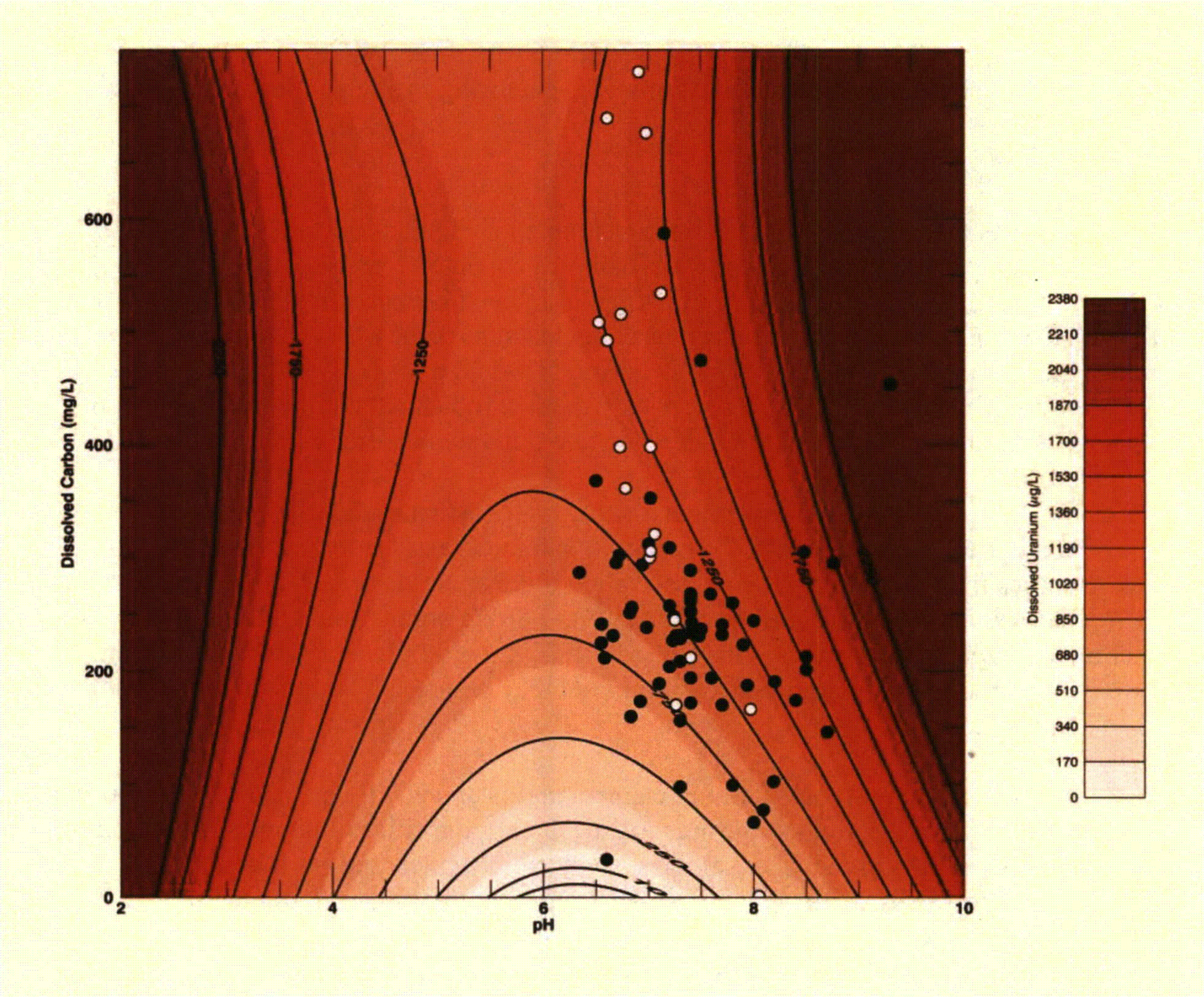

Figure 46. Variation in Uranium Concentrations (µg/L) in Equilibrium with 1 g/L of Hydrous Ferric Oxide and Variable pH and Varying Dissolved Inorganic Carbon Concentrations. *Black and* White *dots* Represent Background *and* Onsite *Bluewater* Groundwater Compositions*,* Respectively.

The diagram in Figure 46 shows uranyl strongly adsorbed at pH values ranging from about 5 to 7.5 with low dissolved carbonate. At pH values less than 4.5, uranium is mostly in the dissolved state regardless of the carbonate concentration. At pH values more than about 8 and modest dissolved carbonate, uranyl is mostly in the dissolved phase. The groundwater samples representing background conditions (background samples) cluster around an area in which about half of the uranium is adsorbed. There are a few groundwater samples that have sufficiently high pH and carbonate concentration that would cause uranyl to strongly partition to the aqueous phase. The onsite samples (white dots on Figure 46) generally have higher dissolved carbonate than the background samples and adsorb less uranyl.

Although Figure 46 was developed specifically for $Fe(OH)_3$, it can be used to generalize the degree to which uranyl adsorption will occur in either the alluvial or San Andres aquifers in response to changes in pH and carbonate concentration. These geochemical modeling results indicate that, if adsorbent minerals are present in the aquifers, uranyl will likely be partially adsorbed. However, short of having information regarding the actual concentrations of adsorptive minerals in the subsurface within the Grants-Bluewater Valley, it is difficult to conclude whether significant adsorption of uranyl is occurring in site and regional groundwater.

7.2.2.2 Mineral Saturation

Saturation indexes for uranium minerals were calculated using the PHREEQC code (Parkhurst and Appelo 2013) for groundwater samples collected from six onsite wells in May 2013. The samples examined were of particular interest because they had the highest uranium concentrations out of all the samples collected at the time of the sampling. The resulting saturation indexes were used to determine if a mineral would dissolve in site groundwater or precipitate from it. As previously stated, a negative index indicated that the mineral would dissolve, whereas a positive saturation index indicated that the mineral would precipitate.

Saturation indexes are logarithmic such that a value of −1 indicates that the groundwater is an order-of-magnitude undersaturated with respect to that mineral. Due to analytical uncertainties, and uncertainties in thermodynamic data, saturation indexes are not exact. As a general rule, saturation indexes are likely accurate to about 0.5, or half an order-of-magnitude. Both dissolution and precipitation can be rate-limited. Thus, if a groundwater sample has a negative saturation index for a particular mineral, the mineral may be present but dissolving too slowly to reach an equilibrium state.

Because silicon is a common rock-forming element and is universally present in groundwater, saturation indexes were determined for uranophane, a uranium silicate mineral. Dissolved silica was not measured in the groundwater samples, so the silica concentration was set at saturation with respect to amorphous silica (SiO_2). The resulting saturation indexes suggested that all six groundwater samples were undersaturated with uranophane by more than 10 orders-of-magnitude (Table 13), indicating that uranophane (and likely all uranyl silicate minerals) is not affecting uranium mobility at the Bluewater site. If equilibrated with quartz instead of amorphous SiO_2, uranophane saturation indexes are about 2 orders-of-magnitude lower.

Table 13. Uranium Mineral Saturation Indexes for Groundwater Samples from Six Bluewater Site Wells That Had the Highest Uranium Concentrations in May 2013

Uranium Mineral	16(SG) [1450][a]	18(SG) [232][a]	I(SG) [334][a]	OBS-3 [228][a]	S(SG) [639][a]	22(M) [380][a]
Gummite UO_3	−9.93	−10.65	−10.58	−10.57	−10.68	−10.57
$Na_4UO_2(CO_3)_3$	−21.92	−24.33	−23.03	−22.39	−22.28	−23.26
Rutherfordine UO_2CO_3	−3.99	−4.90	−4.84	−5.17	−4.87	−4.98
Schoepite $UO_2(OH)_2 \cdot H_2O$	−4.72	−5.43	−5.47	−5.34	−5.48	−5.31
$U(OH)_2SO_4$	−27.42	−26.90	−23.57	−27.88	−20.99	−26.79
U_3O_8	−16.67	−16.89	−13.83	−17.38	−11.33	−15.82
U_4O_9	−33.36	−30.49	−21.26	−32.36	−13.59	−27.81
UO_2(amor)[b]	−15.25	−14.02	−11.12	−14.66	−8.40	−13.11
UO_3(gamma)	−7.18	−7.89	−7.86	−7.81	−7.93	−7.80
Uraninite UO_2(c)[c]	−10.70	−9.51	−6.41	−10.15	−3.85	−8.65
Uranophane $Ca(UO_2)_2(SiO_3OH)_2$	−10.71	−11.80	−12.21	−11.16	−11.80	−11.17

[a] Uranium concentration in micrograms per liter
[b] UO_2(amor) = amorphous UO_2
[c] UO_2(c) = crystalline uraninite

All uranium minerals were undersaturated in all of the groundwater samples (Table 13). In fact, all were undersaturated by at least 3 orders of magnitude, indicating that uranium minerals are not currently precipitating from the groundwater at these locations. These results suggest that uranium mineral precipitation is not an important mechanism affecting transport of uranium in the Bluewater groundwater plumes.

7.2.2.3 Oxidation-Reduction Analysis

Mineral saturation indexes presented in the previous section suggest that chemical reduction potentials sufficient to precipitate reduced uranium minerals are not present at the six sampling locations displayed in Table 13. Because chemical reduction can have a significant effect on uranium mobility, further assessment of possible redox mechanisms was conducted by preparing a compilation of all redox data listed in the DOE environmental database for Bluewater site wells, and then using the data to prepare pE-pH diagrams relative to the speciation of uranium.

Uranium dissolved in groundwater can precipitate low-solubility minerals in water containing chemical reductants, such as organic carbon. Even in a reduced state, uranium contains 2 moles of oxygen per mole of uranium and forms the crystalline mineral uraninite, with the formula UO_2(c). A less crystalline form, amorphous uraninite, with the formula UO_2(amor), is more soluble and more likely to form at shallow groundwater temperatures than its crystalline counterpart. Reducing conditions formed by oxidation of naturally occurring organic carbon can be sufficient to cause precipitation of UO_2(amor). At near-earth-surface temperatures the rate of precipitation of UO_2(amor) by carbon oxidation is increased by microbial respiration.

The stability field of UO_2(amor) is colored medium blue in the pE-pH diagram presented in Figure 47. The predominance fields for dissolved uranium species are shown in lighter color shades. To compile this diagram, the dissolved inorganic carbon concentration was fixed at 120 mg/L.

The pE and pH values for all samples included in the redox analysis are above the stability field for UO_2(amor) (Figure 47). Thus, there is no evidence in the May 2013 samples to suggest that reductive minerals are currently precipitating in the aquifers. Though existing data do not support the presence of redox states low enough to precipitate uranous minerals, redox measurements are problematic due to possible influx of atmospheric oxygen during sampling and analysis. Accordingly, chemically reduced uranium-precipitation zones may be present despite lack of confirmatory data.

Dissolved iron (Fe) concentrations in groundwater were also examined using data from the DOE environmental database. Elevated concentration of dissolved Fe at near-neutral pH is an indicator of chemical reduction because oxidized Fe minerals have a low solubility. Figure 48, which shows the stability fields for amorphous $Fe(OH)_3$ and ferrous carbonate (siderite) at a dissolved Fe concentration of 1 mg/L, indicates that most of the Bluewater groundwater pH-pE measurements fall within the stability fields for these minerals.

Few data were available for dissolved Fe, and only eight samples had data for both redox parameters and Fe concentrations (Table 14). Of the eight measured Fe concentrations, five were higher than 1 mg/L. The Fe concentrations at E(M), OBS-3, and S(SG) are more than an order of magnitude higher than 1 mg/L. The higher Fe concentrations could indicate that groundwater at the site is not at redox equilibrium, or, alternatively, they could reflect errors in the redox measurements. If the redox measurements are in error, the high Fe concentrations signal a more reduced condition that could affect uranium mobility. It is more likely, however, that some of the elevated Fe concentrations are due to particulate or colloidal Fe that inadvertently was included in the respective groundwater samples. In this case, the Fe results would not be useful for evaluating redox conditions.

Collectively, the analyses of existing redox and Fe concentration data in this section, based on samples of groundwater collected from wells in the vicinity of the Bluewater site, provide little evidence of uranium immobility caused by reductive precipitation.

7.2.3 Uranium Isotopes

The activity ratio (AR) of uranium-234 to uranium-238 (^{234}U to ^{238}U) was used in this study to help identify sources of uranium in groundwater. Values of AR in groundwater at a uranium mill site near Cañon City, Colorado, were used by Zielinski et al. (1997) to help distinguish between dissolved uranium resulting from mill site contamination and naturally occurring background uranium. Samples with AR values more than about 1.3 were attributed to background conditions, and lower values were thought to be from the uranium mill. Other studies provide confirmation that groundwater uranium derived from uranium milling operations has an AR value near 1.0 (e.g., Otton et al. 2010; Kamp and Morrison 2014).

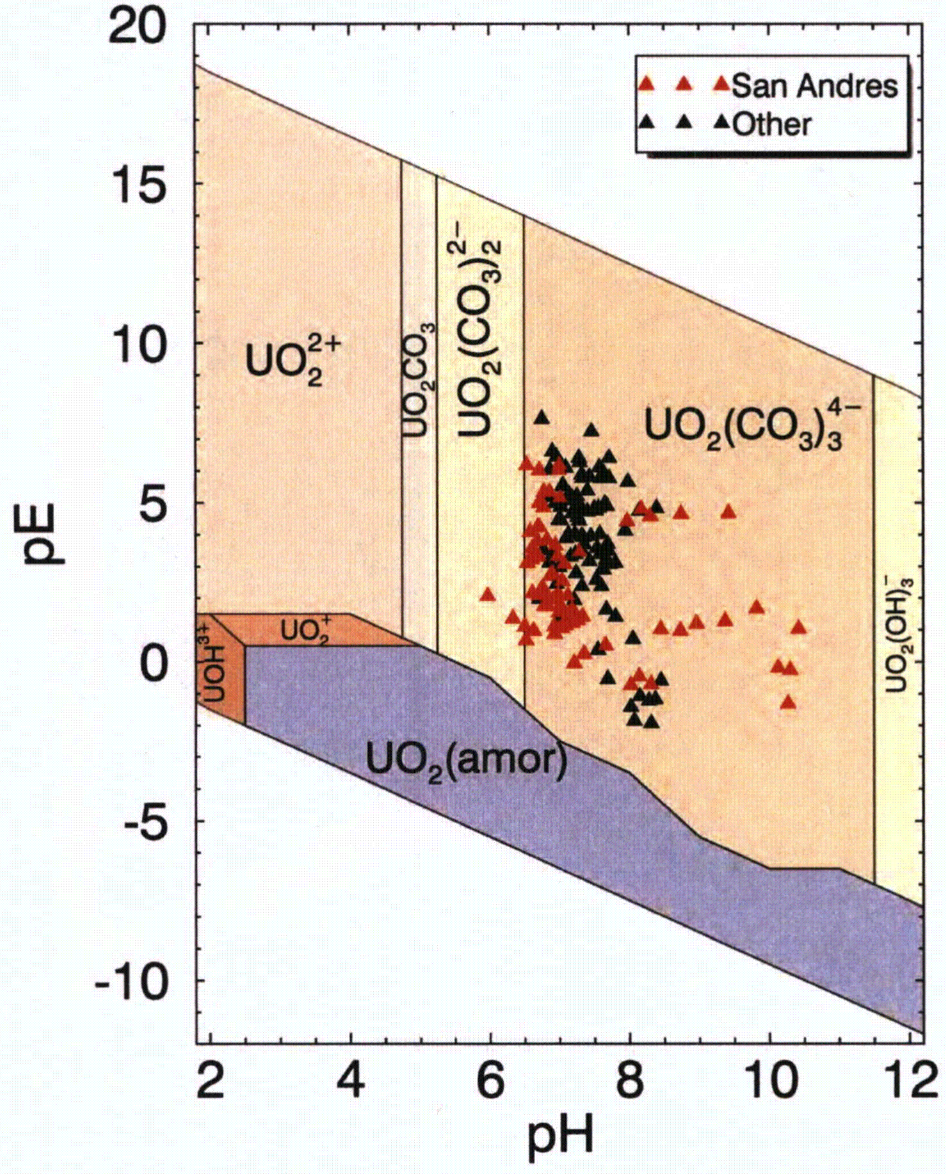

Figure 47. pH-pE Diagram of the U-CO_2 System

System conditions: dissolved carbon = 0.01 mol (120 mg/L), iron = 1.79×10^{-5} mol (1 mg/L), temperature = 25 °C. Sodium nitrate concentrations were held at 0.1 mol for ionic strength control, and sodium concentrations were allowed to fluctuate slightly to maintain charge balance. The mineral stability boundary shown for UO_2(amor) is based on dissolved uranium concentration of 1.85×10^{-7} mol (44 μg/L). The triangles represent individual groundwater samples.

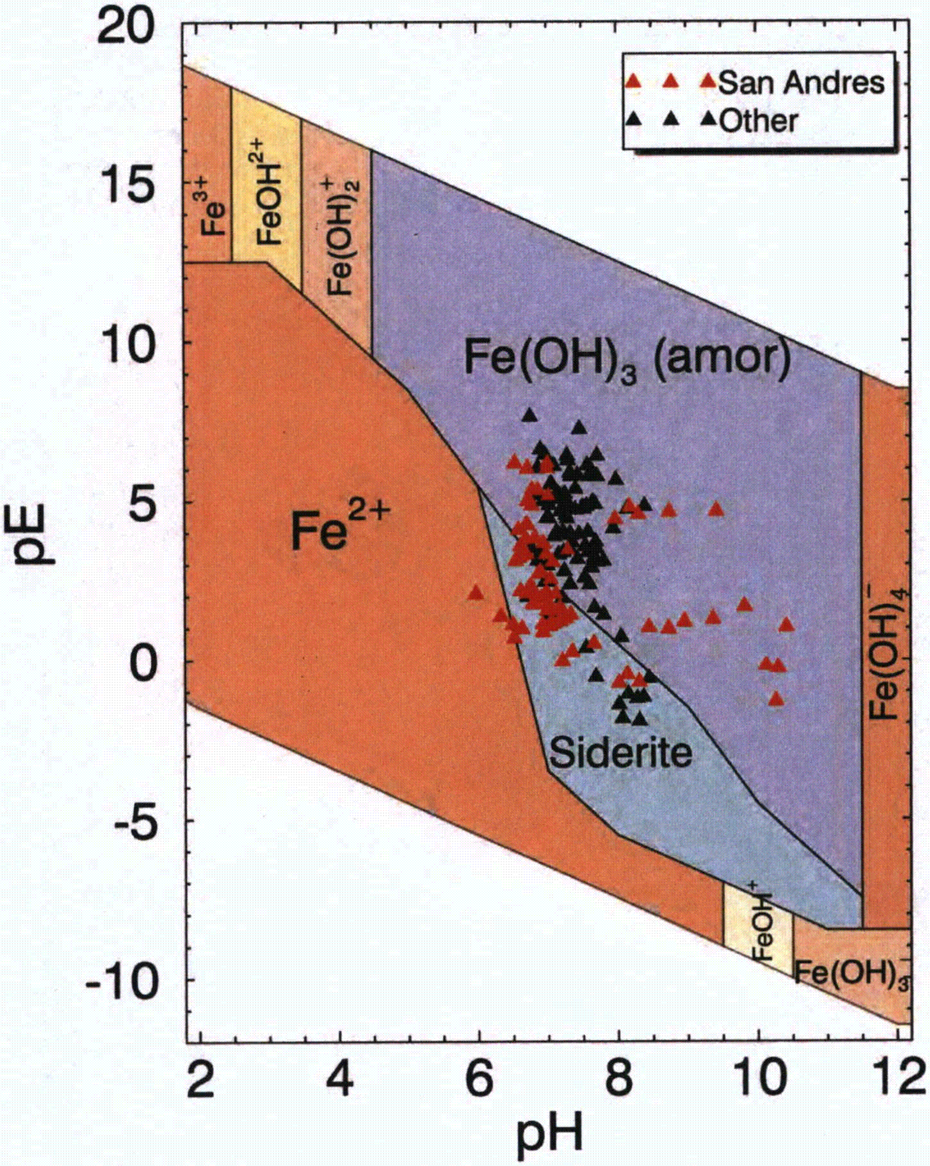

Figure 48. pH-pE Diagram of the Iron System

Conditions are the same as those used to prepare Figure 47. The mineral stability boundaries shown for $Fe(OH)_3$(amor) and siderite are based on a dissolved Fe concentration of 1 mg/L. The triangles represent individual samples of groundwater.

Table 14. Data from DOE Environmental Database for Samples with Reported Concentrations of Dissolved Iron and Redox Parameters

Well	Screened Formation/Aquifer	pH	pE[a]	Iron Concentration (mg/L)[b]
Y2(M)	Quaternary Alluvium	8.14	4.74	0.075
F(M)	Quaternary Alluvium	8.39	4.83	0.44
L(SG)	San Andres Aquifer	9.42	4.67	0.82
I(SG)	San Andres Aquifer	8.74	4.65	1.7
T(M)	Quaternary Alluvium	6.88	5.18	1.7
E(M)	Quaternary Alluvium	7.00	5.01	12
OBS-3	San Andres Aquifer	8.17	4.80	13
S(SG)	San Andres Aquifer	8.30	4.58	570

[a] pE values based on measured oxidation-reduction potential.
[b] All samples were collected on November 10, 2009.

Uranium-234 is produced by the alpha decay of ^{238}U. In a closed system, such as a tight rock matrix, ^{234}U and ^{238}U reach secular equilibrium in a period of about a million years. At secular equilibrium, ^{234}U production from ^{238}U decay is equivalent to its loss through subsequent decay. Under this condition, the AR is 1.0. The uranium in ore deposits, such as those processed at uranium mills like the Bluewater mill, was formed millions of years ago and is near secular equilibrium. Because the milling process uses harsh chemicals to digest the mill feed, the uranium in the mill liquors also has an AR value of near 1.0. In contrast, ^{234}U is preferentially released from uranium-bearing minerals in groundwater aquifers. The preferential release is caused by the disruption of the atomic crystalline structure as ^{234}U is recoiled during the expulsion of an alpha particle from ^{238}U (Cherdyntsev et al. 1955; Kigoshi 1971). The recoiled ^{234}U atom can pass directly into a groundwater pore or may be preferentially leached from its crystalline dislocation. This preferential release of ^{234}U is the cause of elevated AR values that are observed in most natural groundwater.

Uranium isotope data for groundwater wells on the Bluewater site were examined along with isotope information from several other wells in the Grants-Bluewater Valley. In all, 19 samples were collected from 12 wells screened in the alluvial aquifer (Table 15), and 32 samples were collected from 20 wells screened in the San Andres aquifer (Table 16). Data for four municipal wells near Milan, which draw water from the San Andres aquifer, were also included; completion information on the Milan wells was not available. Uranium concentrations measured using chemical methods closely matched concentrations derived from radiometric analyses, confirming the high quality of these data (Table 15 and Table 16). The range in results from chemical analysis of uranium concentrations was 1.1 to 1400 µg/L.

Table 15. $^{234}U/^{238}U$ Activity Ratios (ARs) in Groundwater Samples Collected from Alluvial Aquifer Wells at the Bluewater Site and Surrounding Areas

Well	Alternate Name	Regional Location	Sample Date	Uranium Concentration		Uranium Activity Concentration		Activity Ratio	Data Source
				Chemical (µg/L)	Radiometric (µg/L)	U-234 (pCi/L)	U-238 (pCi/L)		
20(M)		Bluewater Site	11/19/2013	13.9		6.58	4.66	1.41	DOE
21(M)		Bluewater Site	7/27/2011	130	128.5	46.5	43	1.08	DOE
21(M)		Bluewater Site	11/19/2013	137		46.1	40.4	1.14	DOE
22(M)		Bluewater Site	7/27/2011	330	346.7	117	116	1.01	DOE
22(M)		Bluewater Site	11/19/2013	388		122	118	1.03	DOE
23(M)		Bluewater Site	11/19/2013	20.9		8.95	6.24	1.43	DOE
F(M)		Bluewater Site	11/10/2010	8.06	8.8	4.19	2.95	1.42	DOE
F(M)		Bluewater Site	7/28/2011	7.4	7.3	3.4	2.44	1.39	DOE
F(M)		Bluewater Site	11/19/2013	7.34		3.2	2.53	1.26	DOE
T(M)		Bluewater Site	11/9/2010	557	505.2	161	169	0.95	DOE
T(M)		Bluewater Site	7/26/2011	530	544.0	176	182	0.97	DOE
X(M)		Bluewater Site	11/19/2013	145		47.8	44.8	1.07	DOE
Y2(M)		Bluewater Site	11/11/2010	5.19	5.1	2.56	1.7	1.51	DOE
Y2(M)		Bluewater Site	7/28/2011	4.8	4.9	2.61	1.63	1.60	DOE
Y2(M)		Bluewater Site	11/19/2013	5.3		2.97	1.94	1.53	DOE
914	SMC-10	Northeast of Homestake site	3/30/2009	30.9	0.1	0.1	0.04	2.50	NMED
920	SMC-11	Northeast of Homestake site	3/31/2009	228	233.5	78.1	63	1.24	NMED
921	SMC-13	Northeast of Homestake site	4/2/2009	240	192.2	75.8	64.3	1.18	NMED
950	SMC-12	Northeast of Homestake site	3/31/2009	163	155.4	61.9	52	1.19	NMED

µg/L = micrograms per liter; pCi/L = picocuries per liter; DOE = DOE environmental database; NMED = NMED (2010)

Table 16. ^{234}U/^{238}U Activity Ratios (ARs) in Groundwater Samples Collected from San Andres Aquifer Wells at the Bluewater Site and Surrounding Areas

Well	Alternate Name	Regional Location	Sample Date	Uranium Concentration		Uranium Activity Concentration		Activity Ratio	Data Source
				Chemical (µg/L)	Radiometric (µg/L)	U-234 (pCi/L)	U-238 (pCi/L)		
11(SG)		Bluewater Site	11/19/2013	11.7		5.73	3.61	1.59	DOE
13(SG)		Bluewater Site	11/19/2013	98.5		37.8	35.4	1.07	DOE
14(SG)		Bluewater Site	11/19/2013	74.1		26.9	23.8	1.13	DOE
15(SG)		Bluewater Site	11/19/2013	174		60.2	55.8	1.08	DOE
16(SG)		Bluewater Site	11/19/2013	1400		381.0	401.0	0.95	DOE
18(SG)		Bluewater Site	11/19/2013	127		44.7	44.1	1.01	DOE
I(SG) (235 ft)		Bluewater Site	11/11/2010	2.7	4.2	1.48	1.4	1.06	DOE
I(SG) (235 t)		Bluewater Site	7/27/2011	1.1	1.3	0.476	0.449	1.06	DOE
I(SG) (210 ft)		Bluewater Site	11/19/2013	346		53.4	54.0	0.99	DOE
I(SG) (240 ft)		Bluewater Site	11/19/2013	149		49.9	49.1	1.02	DOE
I(SG) (265 ft)		Bluewater Site	11/19/2013	334		110	106	1.04	DOE
I(SG) (300 ft)		Bluewater Site	11/19/2013	324		105	103	1.02	DOE
L(SG)		Bluewater Site	7/27/2011	3.2	3.3	1.85	1.11	1.67	DOE
L(SG)		Bluewater Site	11/19/2013	2.94		1.33	1.2	1.11	DOE
S(SG)		Bluewater Site	7/26/2011	260	231.4	71	77.4	0.92	DOE
S(SG)		Bluewater Site	11/20/2013	525		163	176	0.93	DOE
OBS-3 (255 ft)		Bluewater Site	11/10/2010	1.1	1.4	0.422	0.456	0.93	DOE
OBS-3 (325 ft)		Bluewater Site	11/10/2010	0.648	1.7	0.526	0.558	0.94	DOE
OBS-3 (152 ft)		Bluewater Site	7/28/2011	120	119.0	37.2	39.8	0.93	DOE
OBS-3		Bluewater Site	11/20/2013	9.31		3.05	2.56	1.19	DOE
HMC-951	BW-34	Bluewater Site	8/27/2008	53.3	36.8	13.5	12.3	1.10	NMED
HMC-951	BW-34	Bluewater Site	11/20/2013	31.3		12.0	11.0	1.09	DOE
Bowlins DQ #2	BW-05	West of Bluewater Site	8/25/2008	10.5	9.0	6.4	3	2.13	NMED
BW-14		West of Bluewater Site	8/27/2008	10.5	10.2	13.8	3.4	4.06	NMED
Bowlins	BW-24	West of Bluewater Site	8/25/2008	10.9	9.6	14.4	3.2	4.50	NMED
911	BW-15	Southeast of Bluewater Site	8/25/2008	12	8.4	4.5	2.8	1.61	NMED
949	BW-23	Southeast of Bluewater Site	8/25/2008	13.8	12.9	7.1	4.3	1.65	NMED
928	BW-32	North of Bluewater Site	9/16/2008	29	32.9	22.9	11	2.08	NMED
B-23	Milan Well #1	Milan Municipal Well	12/2/2009	4	3.9	3.07	1.3	2.36	NMDWB
B-35	Milan Well #3	Milan Municipal Well	6/19/1996	13	11.0	6.26	3.69	1.70	NMDWB
B-35	Milan Well #3	Milan Municipal Well	12/2/2009	4	3.7	2.6	1.25	2.08	NMDWB
B-50	Golden Acres, Milan Well #4	Milan Municipal Well	12/2/2009	12	11.4	5.79	3.81	1.52	NMDWB

µg/L = micrograms per liter; pCi/L = picocuries per liter; DOE = DOE environmental database; NMDWB = New Mexico Drinking Water Bureau; NMED = NMED (2010)

Figure 49 contains posted values of AR and uranium concentration for the most recent samples collected at wells in the alluvial aquifer. The ARs for at least four alluvial aquifer wells at the Bluewater site (T(M), X(M), 21(M), and 22(M)) are likely impacted by tailings-related uranium contamination. These four wells, with posted uranium concentrations varying between 130 and about 560 µg/L and posted ARs ranging from 0.95 to 1.08 (Figure 49), appear to lie along the flow path following the ancestral Rio San Jose paleochannel extending southeast from the southeast corner of the Bluewater site (Section 4.3.3 and Figure 21). It is also likely that the sample collected from alluvial aquifer well 23(M) at the Bluewater site contained some mill-related uranium contamination. Though the AR for this well is 1.43, the corresponding uranium concentration is about 21 µg/L, about double the adopted background uranium concentration of 10 µg/L (Section 5.3). In addition, inspection of mapped hydraulic heads and flow directions in the alluvial aquifer in 2012 (Figure 21 and Figure 22) indicates that well 23(M) also lies within the paleochannel extending southeast from the Bluewater site. The combination of an elevated uranium concentration with an AR distinctly larger than 1 suggests that mill-related uranium contamination is present in groundwater at this well location, but its impact has been diluted by mixing with uncontaminated water.

Wells F(M) and Y2(M) at the Bluewater site do not appear to be impacted by mill-related uranium. ARs at these locations range from 1.26 to 1.6, and corresponding uranium concentrations are all less than 10 µg/L (Table 15 and Figure 49). Interpretation of the uranium isotope data for well 20(M) is less clear. Though the single calculated AR for the well is 1.41, the corresponding uranium concentration is about 14 µg/L.

Three of the four wells with uranium isotope data in the alluvial aquifer north-northeast of the Homestake site (Figure 49) have computed ARs ranging from 1.18 to 1.24. These values, when combined with corresponding uranium concentrations between 163 and 240 µg/L, suggest that tailings-related uranium contamination is present in the San Mateo Creek alluvium upgradient of the Homestake site. The source of this contamination is presumably mining- and milling-impacted water from Ambrosia Lake Valley. Well 914, with an AR of 2.5, also appears to be impacted by mill-related uranium contamination. The corresponding uranium concentration at this location is about 31 µg/L, which is greater than the assumed background concentration for uranium of 10 µg/L.

A map with posted uranium isotope ARs for San Andres aquifer wells (Figure 50) further illustrates how the activity ratios can be used to help distinguish tailings-related uranium contamination at the Bluewater site from either uncontaminated groundwater or groundwater that represents a mixture of mill-related, dissolved uranium and background water. The posted ARs at four site wells directly east of, and hydraulically downgradient of, the main tailings disposal cell (OBS-3, S(SG), 16(SG), and I(SG)) range from 0.93 to 0.99, and corresponding concentrations range from 346 to 1400 µg/L. Uranium isotope data for onsite San Andres aquifer wells located south of the main tailings disposal cell (13(SG), 14(SG), 15(SG), 18(SG)) are also representative of mill-related impacts. AR values at these locations vary from 1.01 to 1.13, and corresponding uranium concentrations range from about 74 to 174 µg/L (Table 16). Mill-derived uranium contamination also appears to be present at Homestake well HMC-951, which is just east of the Bluewater site's southeast corner. The computed AR at this well is 1.1, and the posted uranium concentration is about 31 µg/L (Figure 50).

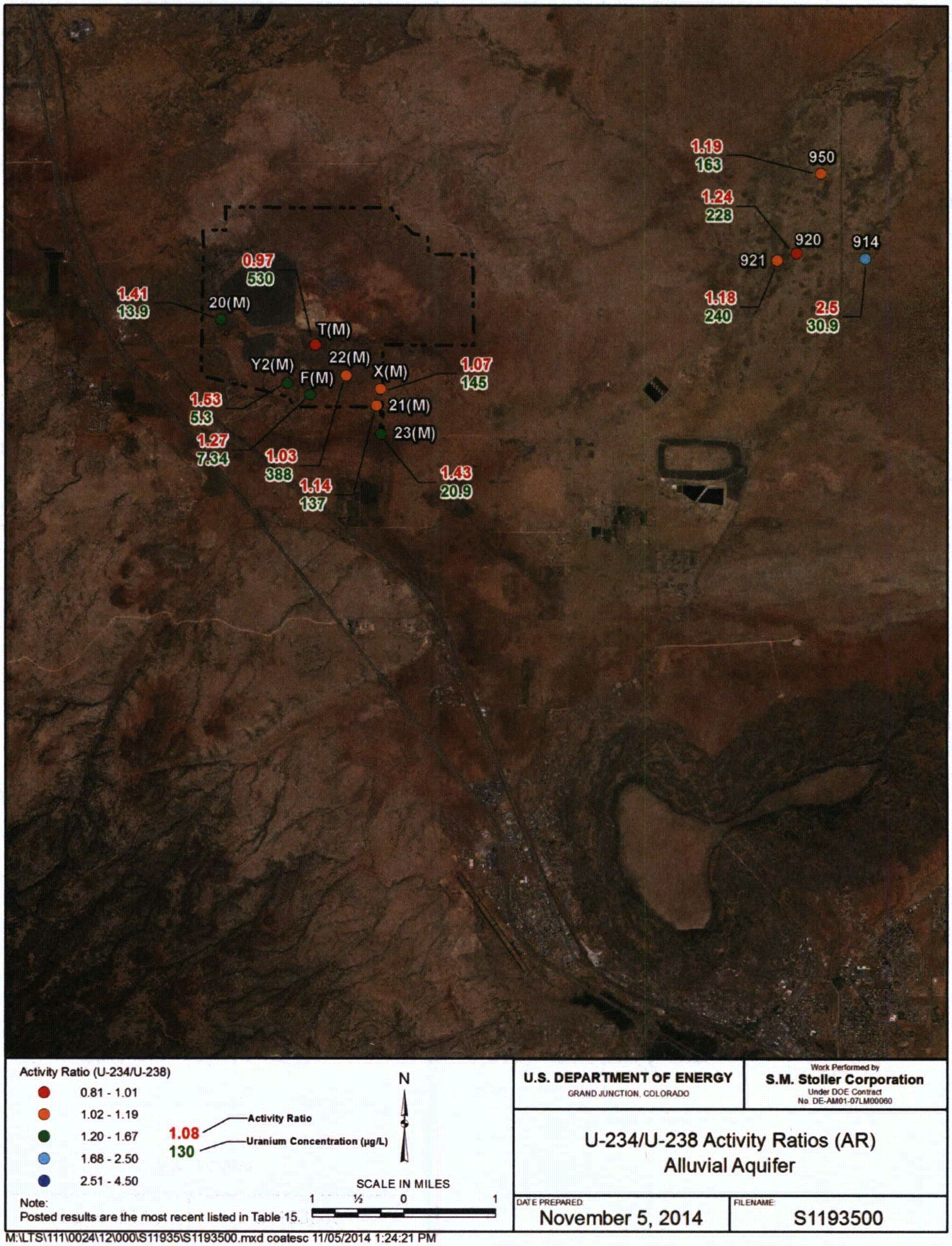

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Figure 49. Uranium Isotope Activity Ratios (ARs) for Alluvial Aquifer Wells at the Bluewater Site and in Surrounding Areas

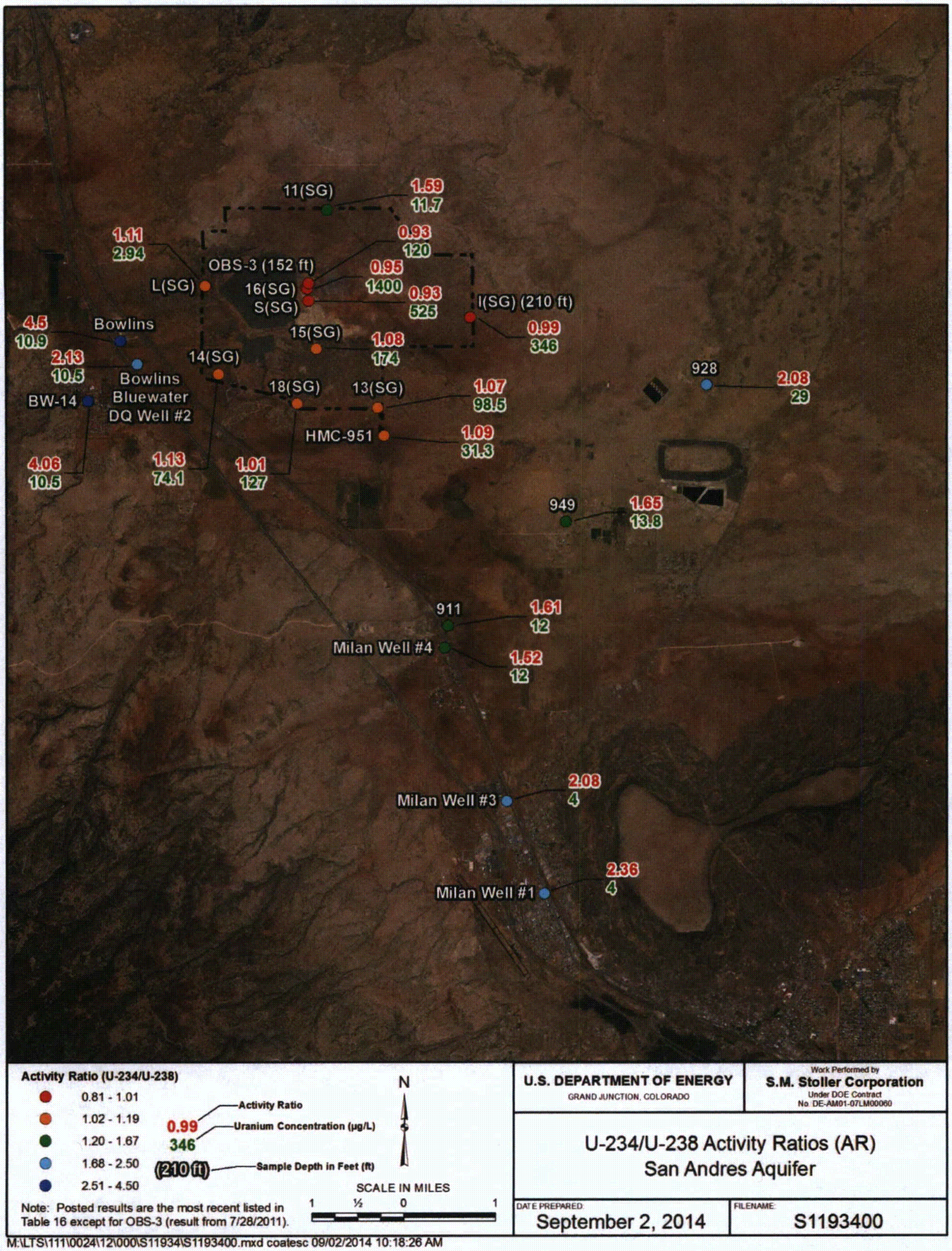

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Figure 50. Uranium Isotope Activity Ratios (ARs) for San Andres Aquifer Wells at the Bluewater Site and in Surrounding Areas

Well 11(SG), located north of the main tailings disposal cell and near the Bluewater site's north border, appears to be unaffected by tailings-derived uranium. The single AR value for the well is 1.59, and the associated uranium concentration is about 12 µg/L, which could be representative of background uranium levels farther north in the aquifer. Similarly, no evidence of uranium contamination is observed at onsite well L(SG), with a calculated AR of 1.11 and a uranium concentration of about 3 µg/L. L(SG) is the background well for the San Andres aquifer at the Bluewater site (Figure 50).

The ARs at three wells west of and upgradient of the Bluewater site (Bowlins, Bowlins DQ#2, and BW-14) range from 2.13 to 4.5 (Figure 50), which are considered representative of groundwater that contains naturally occurring uranium. Of some interest is the fact that the corresponding uranium concentrations at these three wells, varying from 10.5 to 10.9 µg/L, are all essentially equal to the adopted background concentration of 10 µg/L.

The results presented in Table 16 for two I(SG) samples merit further analysis. Both of the I(SG) samples, collected at a depth of 235 ft in 2010 and 2011, had low uranium concentrations (2.7 and 1.1 µg/L, respectively), yet the associated ARs were near a value of unity, indicating the presence of tailings-related contamination. This set of illogical results can be explained by the fact that low-flow sampling techniques were employed to collect samples at the well in 2010 and 2011 and that concentrations in the well bore vary with depth, a topic that was briefly addressed in Section 7.2.1.2. Chapter 8 provides further discussion of the effects of low-flow sampling at well I(SG).

Two OBS-3 samples, collected at depths of 255 and 325 ft in 2010, also exhibit the illogical combination of low uranium concentration and low ARs near the value of unity (Table 16). Because uranium contamination stemming from leached tailings at the main disposal cell has historically been detected at this well and at other wells directly east of the disposal cell, an AR of about 1 is feasible, but the low uranium concentrations are probably attributable to other phenomena that influence this well. As discussed later in Chapter 8, the illogical results seen at this well can be attributed to a combination of severe corrosion of the well screen in well OBS-3 and the use of low-flow sampling techniques.

Two San Andres aquifer wells in the general vicinity of the Homestake site (928 and 949 in Figure 50) show computed ARs of 1.65 and 2.08. These results, suggesting that neither well is directly impacted by tailings-related uranium, could be representative of San Andres aquifer groundwater that is a mixture of uranium-contaminated and uncontaminated waters. This possibility is supported by corresponding uranium concentrations at wells 928 and 949 of 29 and 13.8 µg/L, respectively (Table 16). Though these concentrations are not particularly high, they are, nevertheless, higher than the adopted background uranium concentration.

The northernmost drinking-water supply well for Milan (Milan Well B-50) and nearby well 911 have computed ARs of 1.52 and 1.61, respectively. These values suggest that neither well is directly impacted by tailings-related contamination. However, the corresponding uranium concentrations of 12 µg/L at both wells (Table 16) are slightly higher than the background concentration of 10 µg/L. Thus, the possibility that the water sampled at these two wells represents a mixture of uranium-contaminated and uncontaminated groundwater cannot be completely discounted. A discussion of regional groundwater flow directions in the San Andres aquifer in Section 4.3.3.1 indicates that the two wells lie far south of the flow paths for groundwater that might be contaminated at the Bluewater site and that subsequently migrates to the east-southeast in the direction of Grants. As discussed later in the next chapter, alternative mechanisms of delivering uranium-impacted groundwater to this part of the San Andres aquifer are worthy of examination.

The combination of a computed AR of 2.36 and an associated uranium concentration of 4 µg/L at Milan well B-23 (Figure 50 and Table 16) suggests that this southernmost San Andres aquifer well for Milan remains unaffected by uranium-contaminated groundwater in the Grants-Bluewater Valley. However, the results for Milan well B-35 are somewhat mixed. The combination of a computed AR of 1.70 and a uranium concentration of 13 µg/L in a sample collected in 1996 leave open the possibility that San Andres-aquifer groundwater in the vicinity of the well may have at one time been slightly impacted by uranium contamination. Alternatively, a uranium concentration of 4 µg/L and a computed AR of 2.08 in a sample collected in 2009 imply that the well was not impacted. Such temporally variable results suggest that possible mechanisms for delivering uranium-contaminated groundwater to municipal wells in the Milan area should be examined.

8.0 Groundwater Contamination and Uranium Transport

8.1 Contaminant Source History

8.1.1 Main Tailings Impoundment

Past investigations of contaminant source loading to the alluvial and San Andres aquifers at the Bluewater site have considered a variety of factors that potentially influenced contamination in the subsurface. Estimates have been made of the rates at which tailings fluids seeped from the bottom of the main tailings impoundment, beginning in 1956 and extending through the mid-1990s (e.g., Applied Hydrology Associates Inc. 1990). Estimates have also been made of the inorganic chemical makeup of source material underlying the impoundment that supplied contaminant mass to groundwater migrating beneath the site (e.g., Hydro-Search 1977). Findings from these assessments indicated that acidic tailings fluids seeping downward from the main tailings impoundment were neutralized, causing contaminants to chemically precipitate and adsorb to aquifer media. This process apparently resulted in a mineralized zone in the geologic materials underlying the disposal cell (basalt, alluvium, limestone, and sandstone).

Temporal plots of contaminant concentrations at key wells located hydraulically downgradient of the impoundment have shown that constituent concentrations in the subsurface have remained relatively constant since the early 1980s. This suggests that the rate of contaminant mass loading to the aquifers has remained relatively constant or has been decreasing at a very slow rate over the past 30 years. There is no evidence in the temporal histories to indicate that a new pulse of contamination from the disposal cell has taken place (DOE 2014).

Applied Hydrology Associates Inc. (1995) suggested that contaminant concentrations at downgradient wells were gradually declining prior to decommissioning of the Bluewater site in the 1990s. Trend analyses of the data they examined indicated that, if concentrations were declining, the rate of decrease was extremely slow. Unfortunately, much of the concentration data collected at the site between the mid-1990s and 2011 were largely insufficient for detecting trends. However, a limited quantity of uranium concentration data collected at new wells installed in 2011 and 2012 indicate that rates of contaminant mass loading to the aquifers are either constant or slightly decreasing.

Multiple processes have been proposed for how contaminant mass is mobilized in groundwater beneath the main tailings disposal cell. These include desorption of contaminants from the surfaces of grains that compose alluvial aquifer and San Andres aquifer media and dissolution of contaminants co-precipitated with minerals formed during neutralization of tailings fluids. Alternatively, it is possible that some aqueous-phase contamination continues to reside in low-permeability materials in the alluvial and San Andres aquifers, and it is slowly being released to more-permeable media. Though the nature of the contaminant mobilization is not thoroughly understood, the phenomena involved appear to be uninterrupted.

8.1.2 Anaconda Injection Well

As discussed in Chapters 2 and 6, sandstone strata within the Yeso Formation were used for injection of Bluewater-mill waste fluids from 1960 to 1977, when Anaconda delivered the fluids to a disposal well about 1 mile northeast of the main tailings impoundment. It has been speculated that some of the injected fluid leaked upward to the San Andres aquifer and has been responsible for uranium contamination observed both historically and currently in San Andres aquifer wells north and east of the main tailing impoundment. San Andres aquifer groundwater in the areas containing these wells migrates offsite in an east-southeast direction toward parts of the GRP north of the large tailings disposal cell at the Homestake site.

The potential for the injection well to be a source of contamination in the San Andres aquifer in areas north of the main tailings impoundment was investigated further by examining historical concentrations of dissolved constituents at the San Andres aquifer well referred to as Monitor, which was located near the injection well (Figure 16, Plate 7). Data included in the DOE environmental database show that, while the well was monitored between the mid-1980s and 1990, uranium concentrations at the Monitor well generally fell in the range of 0.25 to 0.35 mg/L, and TDS concentrations varied from 2,000 to about 2,600 mg/L. The last uranium concentration measured at the Monitor well, from a sample collected in April 1990, was 0.32 mg/L. These values support the assumption that the injection well was a source of contamination in the San Andres aquifer in areas north of the main tailings disposal cell.

Because ARCO decommissioned the Monitor well, it is difficult to discern whether contamination stemming from the injected fluids has continued to feed the uranium plume that originated at the injection well. However, because uranium concentrations at the Monitor well were as high as 0.32 mg/L in 1990, some 13 years after waste injection ceased, it is plausible that uranium concentrations exceeding the MCL (0.03 mg/L) are still present in the San Andres aquifer near the Anaconda injection well.

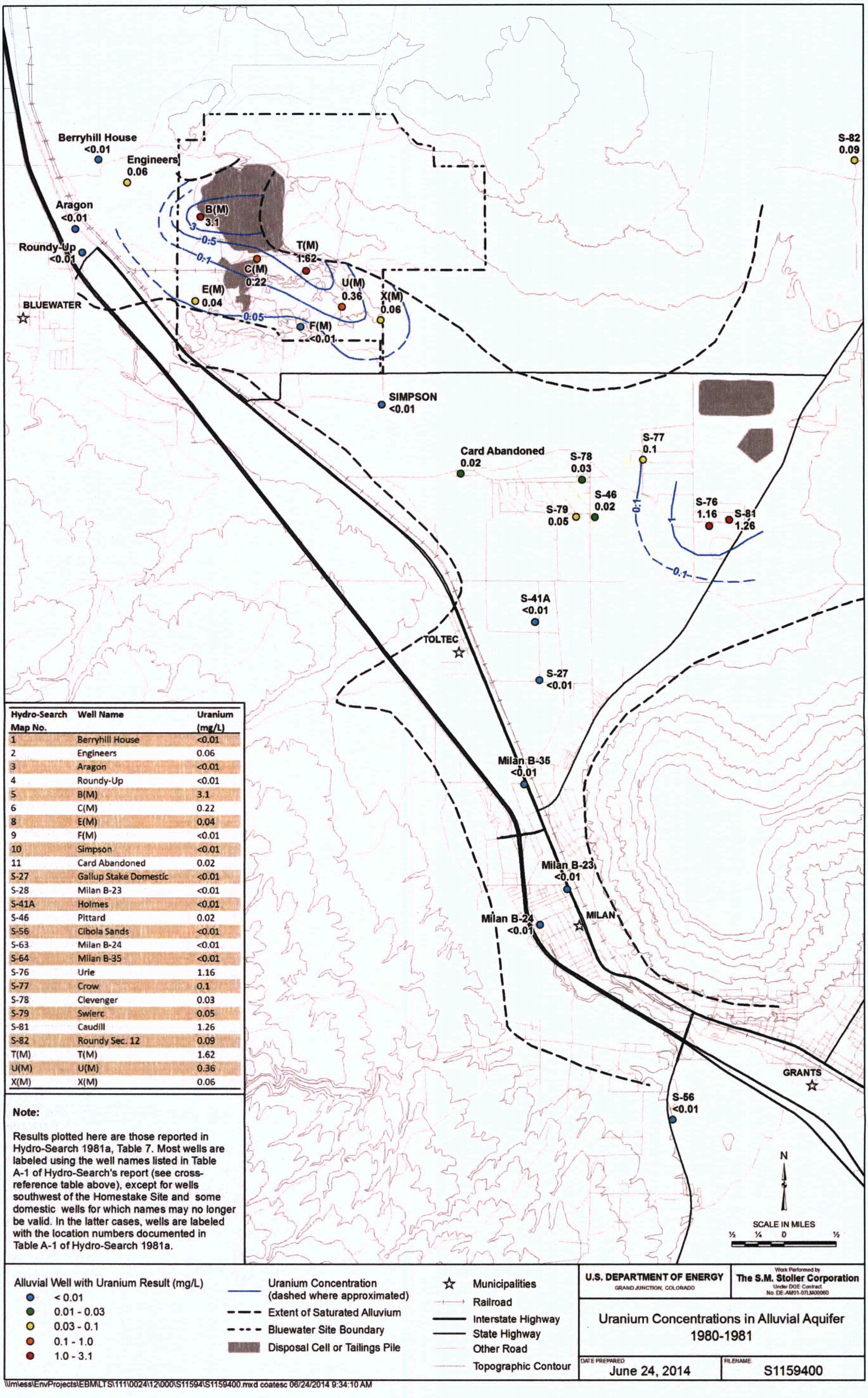
8.2 Uranium Contamination in the Alluvial Aquifer

The spatial extent of uranium contamination originating at the Bluewater site can be partly assessed by comparing historical and recent uranium plume maps in onsite and offsite areas. In addition, examination of the multi-year histories of uranium concentration at the wells used to create the plume maps aerial views of plumes assists in discerning whether contaminant mass is changing or remaining relatively constant in portions of the contamination located hydraulically downgradient of contaminant source areas.

8.2.1 Uranium Plume Maps

To assess how uranium contamination in the alluvial aquifer has evolved over the past 35 years, maps showing uranium concentrations at selected wells and concentration contours in the aquifer were developed for three separate time periods. In addition, a temporal plot of measured uranium concentrations was prepared for five alluvial aquifer wells on the Bluewater site to assess the degree to which steady-state concentrations have been maintained in onsite wells.

Figure 51 shows measured uranium concentrations in 1980–1981 at alluvial aquifer wells that were included in the regional hydrologic study performed by Hydro-Search (1981a). Data coverage at the time was sparse due to a limited number of monitoring wells at both the Bluewater site and the Homestake site. The contoured concentrations in ancestral Rio San Jose alluvium at the Bluewater site are similar to those shown in a comparable map of uranium concentrations prepared by Applied Hydrology Associates Inc. (1990) for conditions in 1982.



Hydro-Search Map No.	Well Name	Uranium (mg/L)
1	Berryhill House	<0.01
2	Engineers	0.06
3	Aragon	<0.01
4	Roundy-Up	<0.01
5	B(M)	3.1
6	C(M)	0.22
8	E(M)	0.04
9	F(M)	<0.01
10	Simpson	<0.01
11	Card Abandoned	0.02
S-27	Gallup Stake Domestic	<0.01
S-28	Milan B-23	<0.01
S-41A	Holmes	<0.01
S-46	Pittard	0.02
S-56	Cibola Sands	<0.01
S-63	Milan B-24	<0.01
S-64	Milan B-35	<0.01
S-76	Urie	1.16
S-77	Crow	0.1
S-78	Clevenger	0.03
S-79	Swierc	0.05
S-81	Caudill	1.26
S-82	Roundy Sec. 12	0.09
T(M)	T(M)	1.62
U(M)	U(M)	0.36
X(M)	X(M)	0.06

Note:

Results plotted here are those reported in Hydro-Search 1981a, Table 7. Most wells are labeled using the well names listed in Table A-1 of Hydro-Search's report (see cross-reference table above), except for wells southwest of the Homestake Site and some domestic wells for which names may no longer be valid. In the latter cases, wells are labeled with the location numbers documented in Table A-1 of Hydro-Search 1981a.

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Uranium Concentrations in Alluvial Aquifer
1980-1981

DATE PREPARED
June 24, 2014

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Figure 51. Regional Uranium Concentrations in the Alluvial Aquifer in 1980–1981

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However, it is difficult to discern from the map the extent of uranium contamination at the time in San Mateo Creek alluvium at the Homestake site. It is very likely that the maximum concentrations in the alluvial aquifer at the Homestake site in 1980 were much larger than the uranium concentrations of 1.16 to 1.26 mg/L posted on Figure 51 for wells directly south of the main tailings disposal cell.

Figure 52 presents a map view of reported uranium concentrations for alluvial aquifer wells in the Grants-Bluewater Valley in 1996–1997. This figure was prepared by combining posted and contoured uranium concentrations from relevant annual GRP reports with concentrations measured at Bluewater site wells during the same time period. Comparison of uranium concentrations in this figure with the uranium concentrations from the 1980–1981 period (Figure 51) reveals that uranium levels in the alluvial aquifer at the Homestake site have indeed been much larger than 1.26 mg/L in past years, and in some cases greater than 50 mg/L. It is difficult to tell from comparing the two maps whether uranium concentrations at the Bluewater site changed significantly between 1980 and 1997. Though concentrations decreased at wells C(M), T(M), U(M), and E(M), apparent increases in concentration were seen at wells B(M) and X(M).

Figure 52 shows two lobes of uranium contamination extending to the south toward Milan, with the uranium concentration in one of the lobes approaching 1 mg/L. The west lobe represents the combined uranium input from the alluvial aquifer plume originating at the Bluewater site and the Homestake plume extending westward from the large tailings disposal cell. It is impossible to tell from this figure whether the location of the leading edge of each of the contaminant lobes, as defined by a concentration of 0.01 mg/L, may have extended as far south as the Milan municipal wells in years preceding 1996–1997. However, this possibility cannot be entirely dismissed given the assessment of uranium concentrations and uranium isotope ratios for Milan Wells B-35 and B-50 in Section 7.2.3.

The potential for uranium contamination in the alluvial aquifer extending as far south as some of the Milan municipal wells at some time in the past can be further assessed by examining the potentiometric surface for the alluvial aquifer in fall 2012 (see Figure 21 and Figure 22), which indicates that a hydraulic sink has occurred in the aquifer during recent years in an area just to the east and northeast of Toltec. This finding, combined with the observation in this study that the alluvial aquifer directly overlies the San Andres Limestone in an area extending from about a mile north of Toltec to Grants (see Figures 18, 19, and 52), suggests that downward migration of groundwater from the alluvial aquifer to the San Andres aquifer in response to pumping at Milan Wells B-35 and B-50 is feasible. This is especially true given that the groundwater sink has been observed near the southern tip of the west lobe of uranium contamination formed by merging of uranium plumes from the Bluewater and Homestake sites (Figure 52).

Figure 53 presents a map view of uranium concentrations in the alluvial aquifer in 2012–2013. Comparison of this figure with the 1996–1998 sampling results in Figure 52 indicates a sharp decline in uranium concentration near the large tailings disposal cell at the Homestake site over the 16-year span separating the two periods. In addition, an overall decline in uranium concentrations is observed between 1996–1998 and 2012–2013. However, it is difficult to tell from comparing the two maps whether uranium concentrations in the alluvial aquifer at the Bluewater site decreased between the two periods.

Preparation of the uranium plume maps in Figures 52 and 53 revealed the importance of including concentration data from offsite locations in developing the conceptual model for the Bluewater site. Identification of the downgradient extent of the uranium plumes in 1996–1997 and 2012–2013 would not have been possible without using uranium-concentration data from downgradient wells and including plume maps prepared for annual reports on the GRP. This finding suggests that it may be advisable to include offsite data when evaluating the conceptual models for other LM sites.

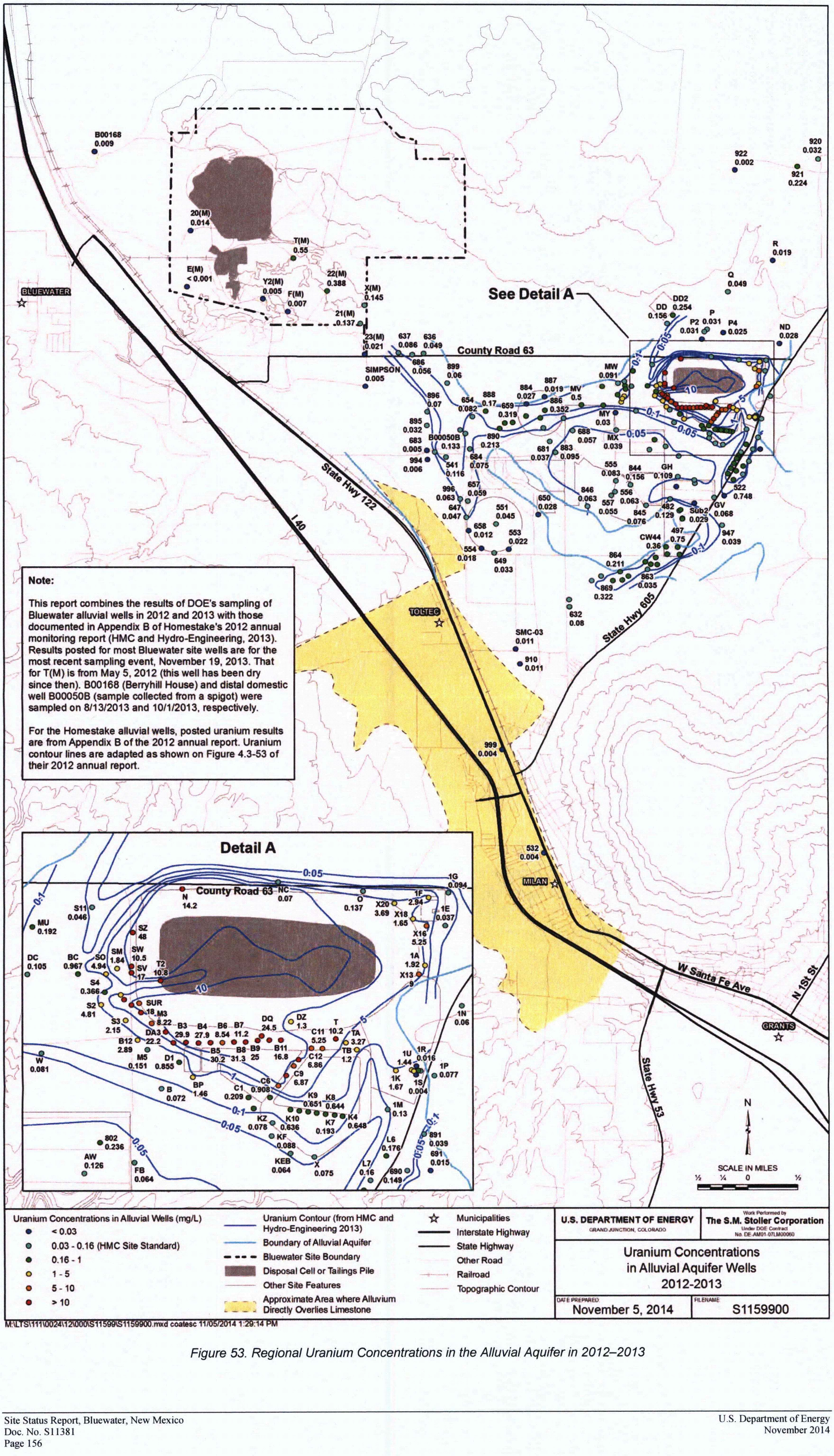
8.2.2 Uranium Concentration Histories

Figure 54 shows the temporal plot of uranium concentrations at alluvial aquifer wells T(M), U(M), X(M), 21(M), and 22(M) between 1984 and 2013. Though the concentrations fluctuate considerably with time, the data, for the most part, suggest that uranium levels in the alluvial aquifer at onsite locations have either remained stable for the past 30 years or have been slowly declining. With the exception of well T(M), the concentration data do not suggest that a pulse of uranium contamination has been migrating from contaminant source areas beneath the Bluewater site.

Prior to 2000, uranium concentrations at well T(M) were relatively steady, remaining within a range of 0.1 to 0.3 mg/L (Figure 54). This was followed by a steady increase in uranium concentration at the well between 2000 and 2010, suggesting that a pulse of uranium contaminant mass had affected the alluvial aquifer. However, analysis of water chemistry data at T(M) and inspection of water levels in the well in recent years indicate that the water being sampled at this location was increasingly from the Chinle Formation and was not representative of the alluvium.

The Savannah River National Laboratory (SRNL 2014) studied the aqueous-phase chemistry of groundwater at well T(M) and concluded that the increase in uranium concentration between 2000 and 2010 was unlikely to have been caused by a pulse release of tailings fluids. SRNL notes that, beginning in the late 1990s, the water table elevation in the alluvial aquifer steadily decreased, and that the water level at well T(M) dropped to the bottom of the 5 ft screened interval in the well in 2008. The bottom of the screen is at the contact between the alluvial aquifer and the Chinle Formation, and the open borehole beneath the screen extends another 9 ft into the Chinle. Thus, virtually all of the water sampled from the well from 2008 to 2012 was from the upper 9 ft of the Chinle, which apparently contained large uranium concentrations.

SRNL (2014) states that the cause of the large uranium concentration in the uppermost part of the Chinle is unclear, but the study provides a possible hypothesis for the responsible chemical processes. The hypothesis is based on assumed diffusion of uranium-impacted groundwater from the alluvial aquifer to the uppermost layers of Chinle Formation over a 30-year period and concomitant interaction of the uranium with minerals in the formation. The diffusion caused either ion exchange of dissolved uranium for other constituents in Chinle Formation rock or precipitation of U(IV) minerals under chemically reducing conditions. Observations made by SRNL supporting the latter mechanism (redox-driven precipitation) include color changes in the uppermost Chinle Formation rock that are considered representative of reducing conditions. In following years, slow remobilization of the uranium in the Chinle Formation rock, perhaps due to the onset of oxidized conditions, led to back diffusion of the contaminant into the formation's pore water. The relatively high concentrations of uranium detected at well T(M) between 2008



Note:

This report combines the results of DOE's sampling of Bluewater alluvial wells in 2012 and 2013 with those documented in Appendix B of Homestake's 2012 annual monitoring report (HMC and Hydro-Engineering, 2013). Results posted for most Bluewater site wells are for the most recent sampling event, November 19, 2013. That for T(M) is from May 5, 2012 (this well has been dry since then). B00168 (Berryhill House) and distal domestic well B00050B (sample collected from a spigot) were sampled on 8/13/2013 and 10/1/2013, respectively.

For the Homestake alluvial wells, posted uranium results are from Appendix B of the 2012 annual report. Uranium contour lines are adapted as shown on Figure 4.3-53 of their 2012 annual report.

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Uranium Concentrations
in Alluvial Aquifer Wells
2012-2013

DATE PREPARED
November 5, 2014

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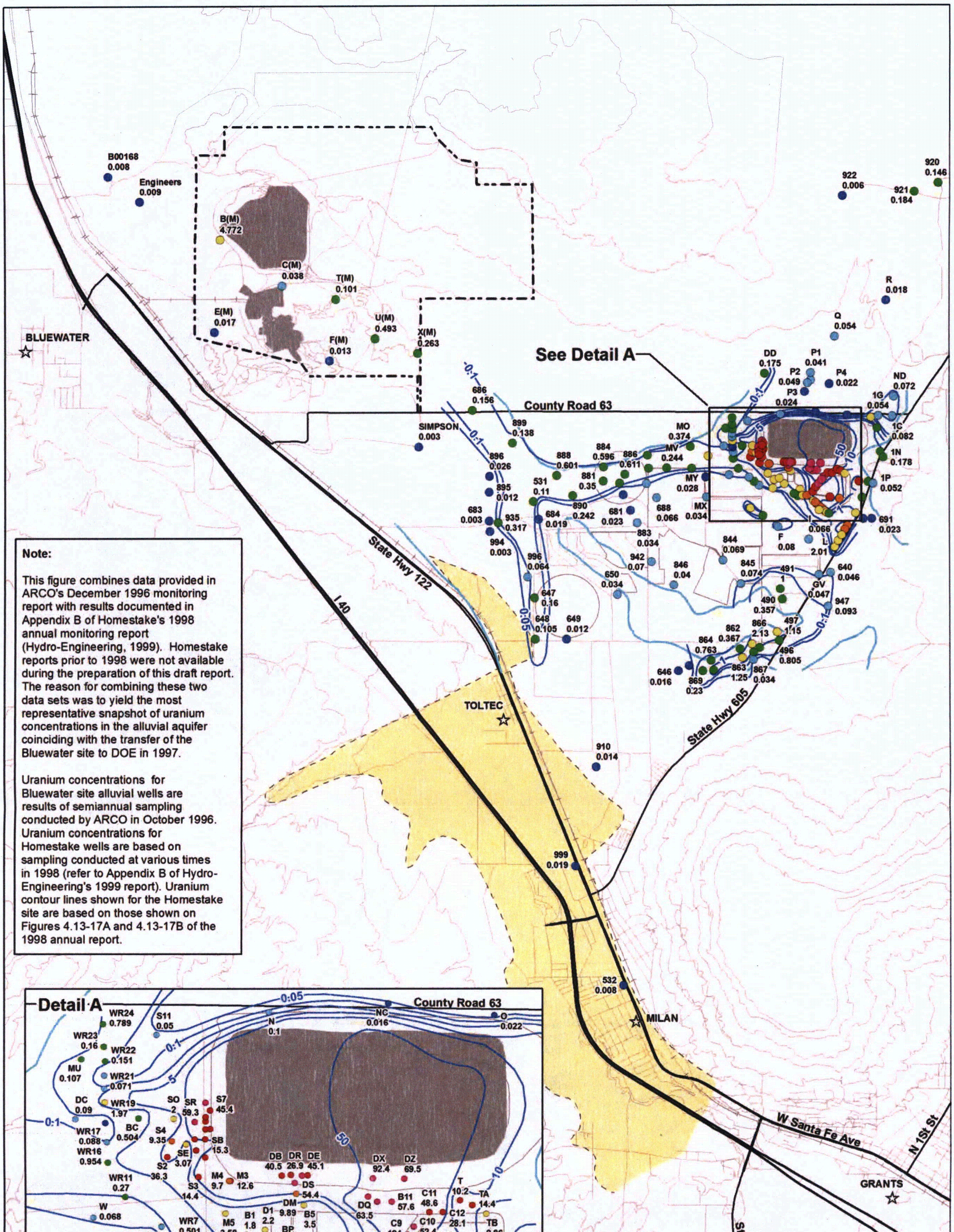
Figure 53. Regional Uranium Concentrations in the Alluvial Aquifer in 2012–2013



Note:

This figure combines data provided in ARCO's December 1996 monitoring report with results documented in Appendix B of Homestake's 1998 annual monitoring report (Hydro-Engineering, 1999). Homestake reports prior to 1998 were not available during the preparation of this draft report. The reason for combining these two data sets was to yield the most representative snapshot of uranium concentrations in the alluvial aquifer coinciding with the transfer of the Bluewater site to DOE in 1997.

Uranium concentrations for Bluewater site alluvial wells are results of semiannual sampling conducted by ARCO in October 1996. Uranium concentrations for Homestake wells are based on sampling conducted at various times in 1998 (refer to Appendix B of Hydro-Engineering's 1999 report). Uranium contour lines shown for the Homestake site are based on those shown on Figures 4.13-17A and 4.13-17B of the 1998 annual report.

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No. DE-AM01-07LM00060

Uranium Concentrations
in Alluvial Wells
1996-1998

DATE PREPARED
November 7, 2014

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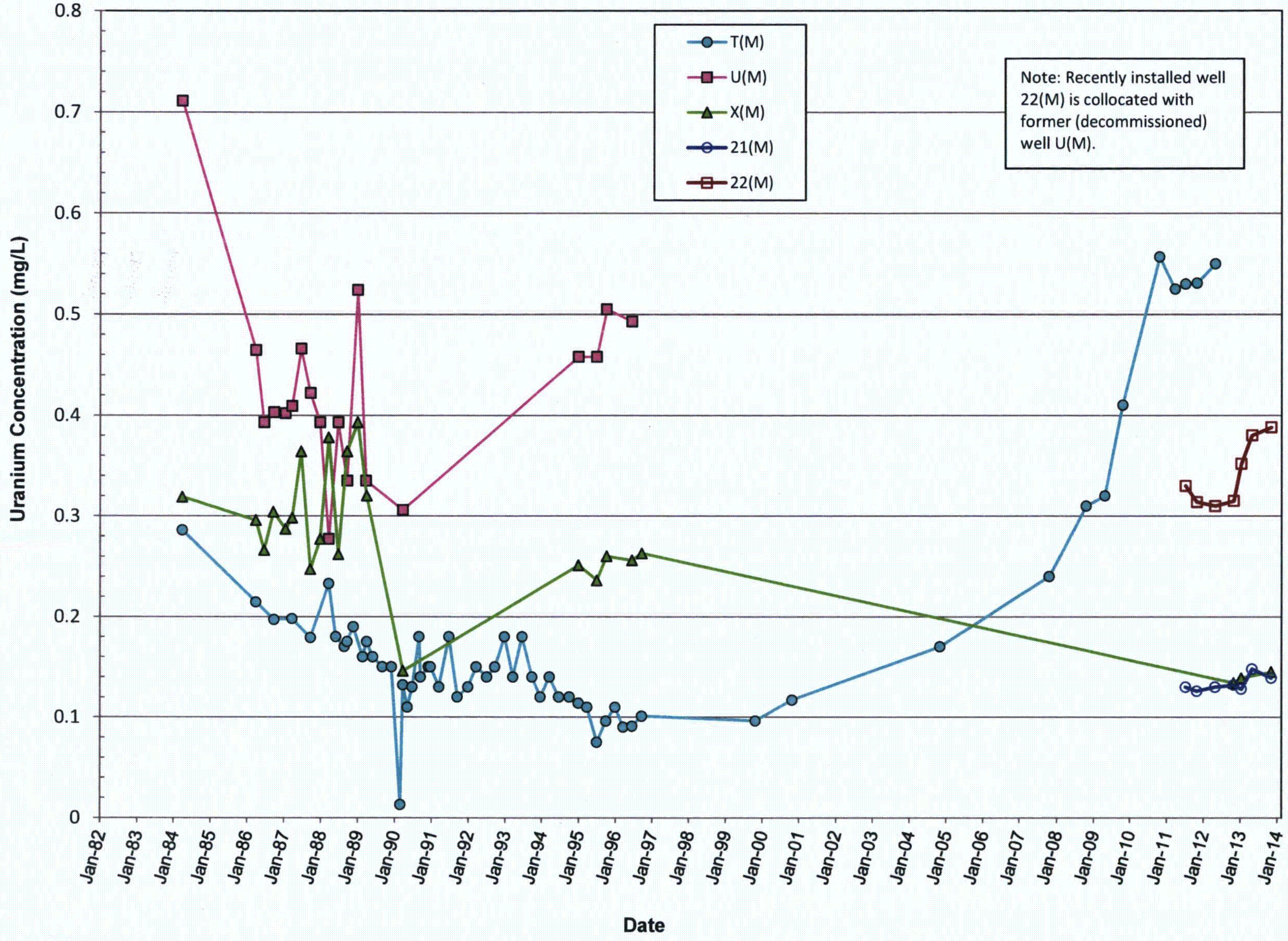
Figure 52. Regional Uranium Concentrations in the Alluvial Aquifer in 1996–1997



Note: Recently installed well 22(M) is collocated with former (decommissioned) well U(M).

Figure 54. Temporal Plot of Uranium Concentrations in Alluvial Aquifer Wells at the Bluewater Site

and 2012 are attributed to the back diffusion (SRNL 2014). The combination of steadily decreasing water levels at T(M) since the late 1990s, increasing proportions of Chinle Formation water in samples collected from the well beginning in 2000, and the findings and hypothesis by SRNL (2014) suggests that increasing uranium concentrations in well T(M) between 2000 and 2010 were not caused by a pulse of uranium contamination migrating southeastward in the alluvial aquifer. Instead, the uranium increases are attributed to increasing amounts of Chinle pore water in samples collected at the well.

Critical assessment of the uranium concentration data at well T(M), as presented above, provides a possible lesson regarding the annual reporting that is performed for many LM sites. Specifically, it may be helpful to further investigate potential causes of increasing and decreasing concentration trends at LM sites, since simple reporting of the concentrations by themselves may fail to capture difficulties with the wells included in the annual monitoring.

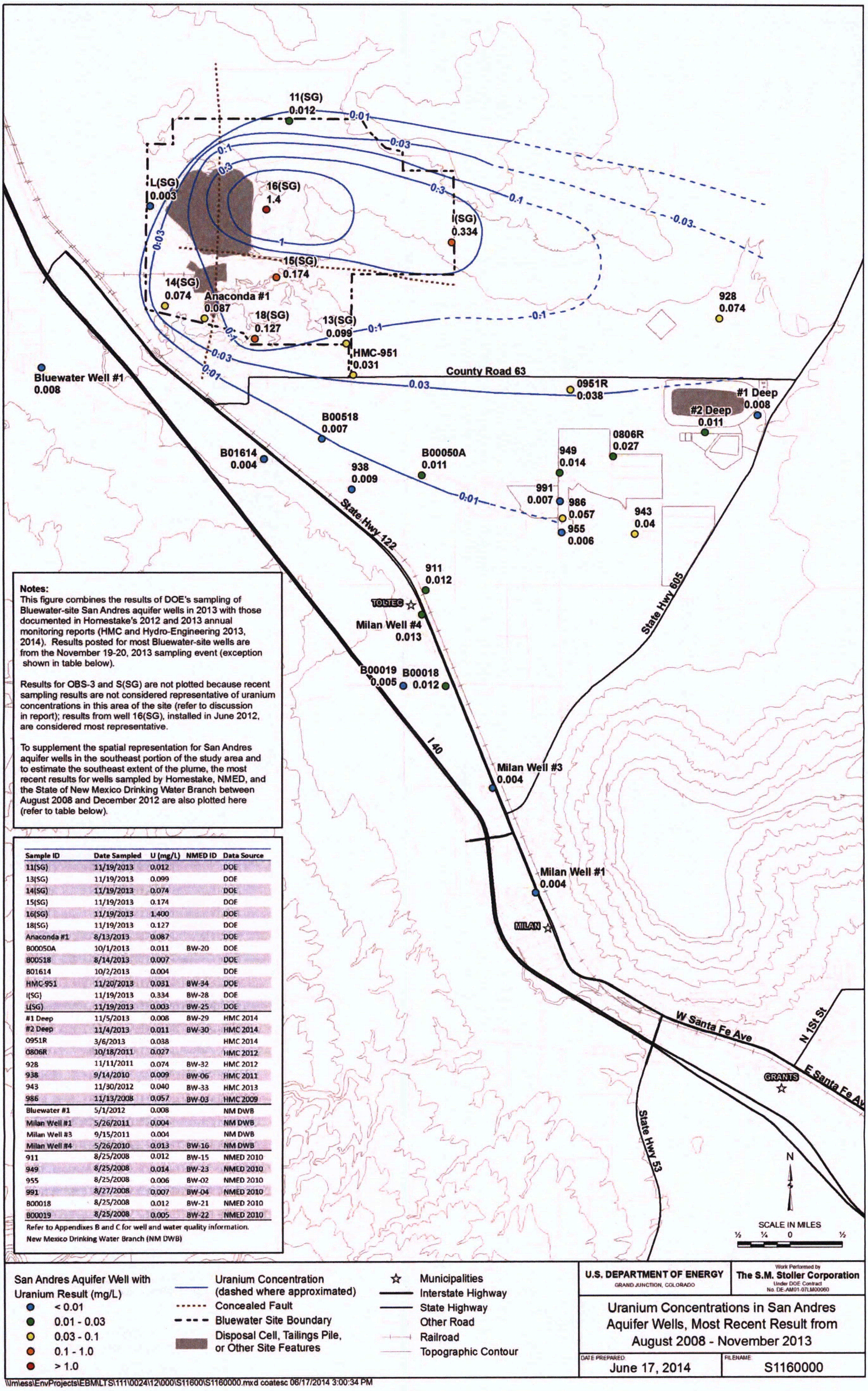
8.3 Uranium Contamination in the San Andres Aquifer

8.3.1 Uranium Plume Maps

Figure 55 presents a map of uranium concentrations measured in 1980 and 1981 at several San Andres aquifer wells in the Grants-Bluewater Valley as part of the Hydro-Search (1981a) assessment of regional hydrogeology and water chemistry. As shown in this figure, the largest uranium concentrations at the time were observed at Bluewater site wells on the east side of the main tailings impoundment, and somewhat elevated concentrations were also observed at the production wells, Anaconda #2 and Anaconda #5.

Of some interest was the relatively high uranium concentration of 0.05 mg/L at well 928 (Roundy Sec. 23 in Figure 55), about 2.5 mi hydraulically downgradient of the Bluewater site and 0.5 mile north of the large tailings disposal cell at the Homestake site. Hydro-Search (1981a) made no attempt to distinguish this latter concentration in its regional assessment, perhaps because the regulatory standard for uranium in 1981 was 5 mg/L, a value that was an order of magnitude higher than the observed concentration at well 928 at the time. It is also possible that the well 928 concentration was considered to be unrelated to the uranium plume at the Bluewater site because Hydro-Search (1977) had estimated that the velocity of groundwater in the San Andres aquifer east of the main tailings impoundment was limited to about 400 ft/yr. If the much higher velocities derived from models by Dames & Moore (1986a) (see Section 4.3.3.2) had been taken into account, arrival of uranium contamination at well 928 from the Bluewater site as of 1980 would have been considered plausible. Such a conclusion would have been supported by the map of ambient flow directions in the San Andres aquifer presented in Figure 24, which indicates that groundwater from various parts of the Bluewater site migrates to multiple locations north and south of the large tailings disposal cell at the Homestake site.

To detect possible changes in the areal extent of uranium contamination since 1980, maps of measured uranium concentrations in the San Andres aquifer were also prepared for three additional periods. Figure 56 through Figure 58 show the maps, which represent the time periods 1996–1997, 2008, and 2012–2013, respectively. As each figure illustrates, the uranium concentrations at Bluewater site wells and areas east of the site were relatively close to concentrations shown for 1980–1981. Similarly, the range of uranium levels (0.03 to 0.07 mg/L) measured in well 928 at the Homestake site was close to the concentration of 0.05 mg/L



Notes:
This figure combines the results of DOE's sampling of Bluewater-site San Andres aquifer wells in 2013 with those documented in Homestake's 2012 and 2013 annual monitoring reports (HMC and Hydro-Engineering 2013, 2014). Results posted for most Bluewater-site wells are from the November 19-20, 2013 sampling event (exception shown in table below).

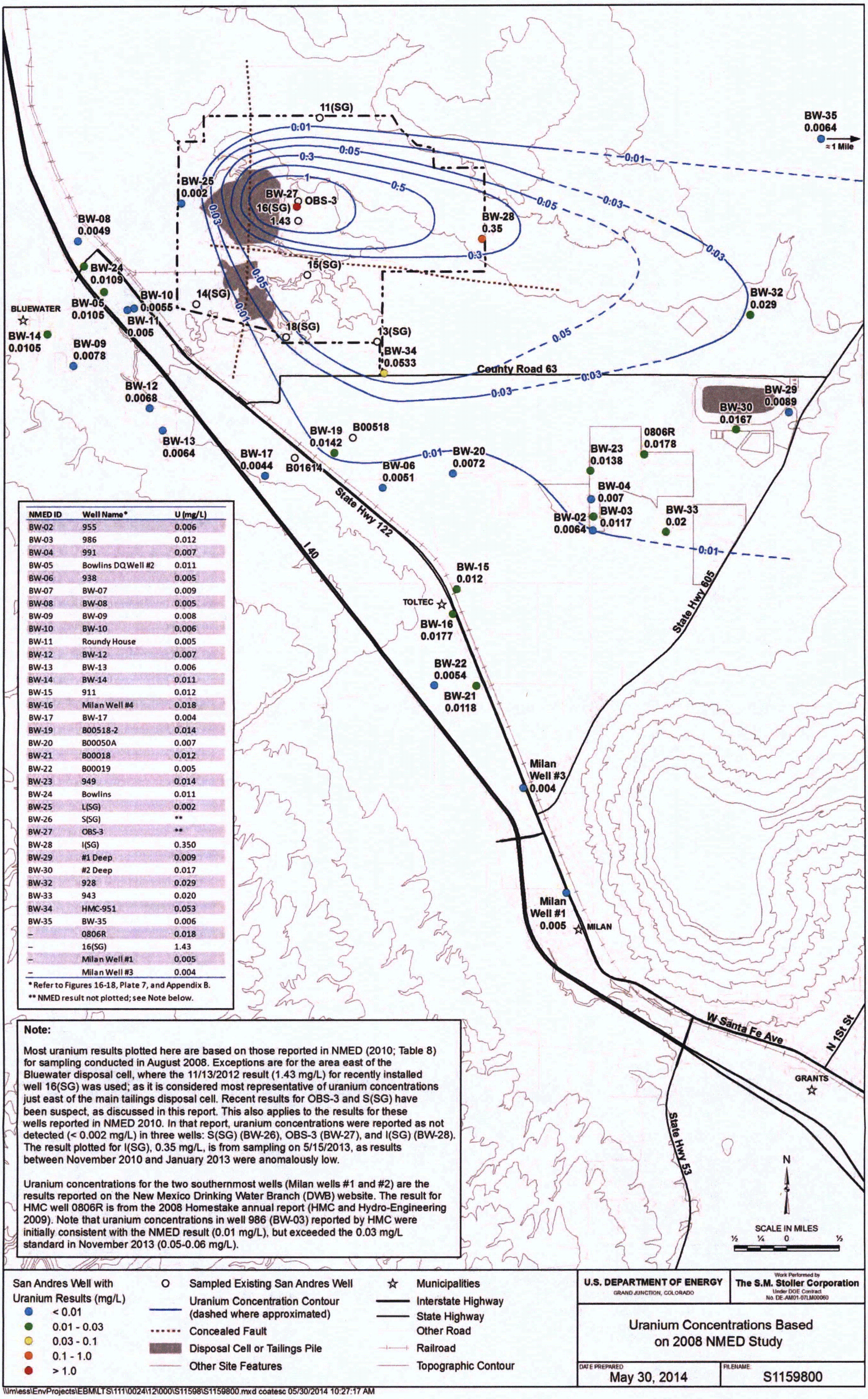
Results for OBS-3 and S(SG) are not plotted because recent sampling results are not considered representative of uranium concentrations in this area of the site (refer to discussion in report); results from well 16(SG), installed in June 2012, are considered most representative.

To supplement the spatial representation for San Andres aquifer wells in the southeast portion of the study area and to estimate the southeast extent of the plume, the most recent results for wells sampled by Homestake, NMED, and the State of New Mexico Drinking Water Branch between August 2008 and December 2012 are also plotted here (refer to table below).

Sample ID	Date Sampled	U (mg/L)	NMED ID	Data Source
11(SG)	11/19/2013	0.012		DOE
13(SG)	11/19/2013	0.099		DOE
14(SG)	11/19/2013	0.074		DOE
15(SG)	11/19/2013	0.174		DOE
16(SG)	11/19/2013	1.400		DOE
18(SG)	11/19/2013	0.127		DOE
Anaconda #1	8/13/2013	0.087		DOE
B00050A	10/1/2013	0.011	BW-20	DOE
B00518	8/14/2013	0.007		DOE
B01614	10/2/2013	0.004		DOE
HMC-951	11/20/2013	0.031	BW-34	DOE
I(SG)	11/19/2013	0.334	BW-28	DOE
L(SG)	11/19/2013	0.003	BW-25	DOE
#1 Deep	11/5/2013	0.008	BW-29	HMC 2014
#2 Deep	11/4/2013	0.011	BW-30	HMC 2014
0951R	3/6/2013	0.038		HMC 2014
0806R	10/18/2011	0.027		HMC 2012
928	11/11/2011	0.074	BW-32	HMC 2012
938	9/14/2010	0.009	BW-06	HMC 2011
943	11/30/2012	0.040	BW-33	HMC 2013
986	11/13/2008	0.057	BW-03	HMC 2009
Bluewater #1	5/1/2012	0.008		NM DWB
Milan Well #1	5/26/2011	0.004		NM DWB
Milan Well #3	9/15/2011	0.004		NM DWB
Milan Well #4	5/26/2010	0.013	BW-16	NM DWB
911	8/25/2008	0.012	BW-15	NMED 2010
949	8/25/2008	0.014	BW-23	NMED 2010
955	8/25/2008	0.006	BW-02	NMED 2010
991	8/27/2008	0.007	BW-04	NMED 2010
B00018	8/25/2008	0.012	BW-21	NMED 2010
B00019	8/25/2008	0.005	BW-22	NMED 2010

Refer to Appendixes B and C for well and water quality information.
New Mexico Drinking Water Branch (NM DWB)

Figure 58. Regional Uranium Concentrations in the San Andres Aquifer for the Period 2008–2013



NMED ID	Well Name*	U (mg/L)
BW-02	955	0.006
BW-03	986	0.012
BW-04	991	0.007
BW-05	Bowlins DQ Well #2	0.011
BW-06	938	0.005
BW-07	BW-07	0.009
BW-08	BW-08	0.005
BW-09	BW-09	0.008
BW-10	BW-10	0.006
BW-11	Roundy House	0.005
BW-12	BW-12	0.007
BW-13	BW-13	0.006
BW-14	BW-14	0.011
BW-15	911	0.012
BW-16	Milan Well #4	0.018
BW-17	BW-17	0.004
BW-19	B00518-2	0.014
BW-20	B00050A	0.007
BW-21	B00018	0.012
BW-22	B00019	0.005
BW-23	949	0.014
BW-24	Bowlins	0.011
BW-25	L(SG)	0.002
BW-26	S(SG)	**
BW-27	OBS-3	**
BW-28	I(SG)	0.350
BW-29	#1 Deep	0.009
BW-30	#2 Deep	0.017
BW-32	928	0.029
BW-33	943	0.020
BW-34	HMC-951	0.053
BW-35	BW-35	0.006
–	0806R	0.018
–	16(SG)	1.43
–	Milan Well #1	0.005
–	Milan Well #3	0.004

* Refer to Figures 16-18, Plate 7, and Appendix B.

** NMED result not plotted; see Note below.

Note:

Most uranium results plotted here are based on those reported in NMED (2010; Table 8) for sampling conducted in August 2008. Exceptions are for the area east of the Bluewater disposal cell, where the 11/13/2012 result (1.43 mg/L) for recently installed well 16(SG) was used; as it is considered most representative of uranium concentrations just east of the main tailings disposal cell. Recent results for OBS-3 and S(SG) have been suspect, as discussed in this report. This also applies to the results for these wells reported in NMED 2010. In that report, uranium concentrations were reported as not detected (< 0.002 mg/L) in three wells: S(SG) (BW-26), OBS-3 (BW-27), and I(SG) (BW-28). The result plotted for I(SG), 0.35 mg/L, is from sampling on 5/15/2013, as results between November 2010 and January 2013 were anomalously low.

Uranium concentrations for the two southernmost wells (Milan wells #1 and #2) are the results reported on the New Mexico Drinking Water Branch (DWB) website. The result for HMC well 0806R is from the 2008 Homestake annual report (HMC and Hydro-Engineering 2009). Note that uranium concentrations in well 986 (BW-03) reported by HMC were initially consistent with the NMED result (0.01 mg/L), but exceeded the 0.03 mg/L standard in November 2013 (0.05-0.06 mg/L).

San Andres Well with Uranium Results (mg/L)
- ● < 0.01
- ● 0.01 - 0.03
- ● 0.03 - 0.1
- ● 0.1 - 1.0
- ● > 1.0

○ Sampled Existing San Andres Well
Uranium Concentration Contour (dashed where approximated)
Concealed Fault
▨ Disposal Cell or Tailings Pile
Other Site Features

☆ Municipalities
Interstate Highway
State Highway
Other Road
Railroad
Topographic Contour

U.S. DEPARTMENT OF ENERGY
GRAND JUNCTION, COLORADO

Work Performed by
The S.M. Stoller Corporation
Under DOE Contract
No. DE-AM01-07LM00060

Uranium Concentrations Based on 2008 NMED Study

DATE PREPARED: May 30, 2014
FILENAME: S1159800

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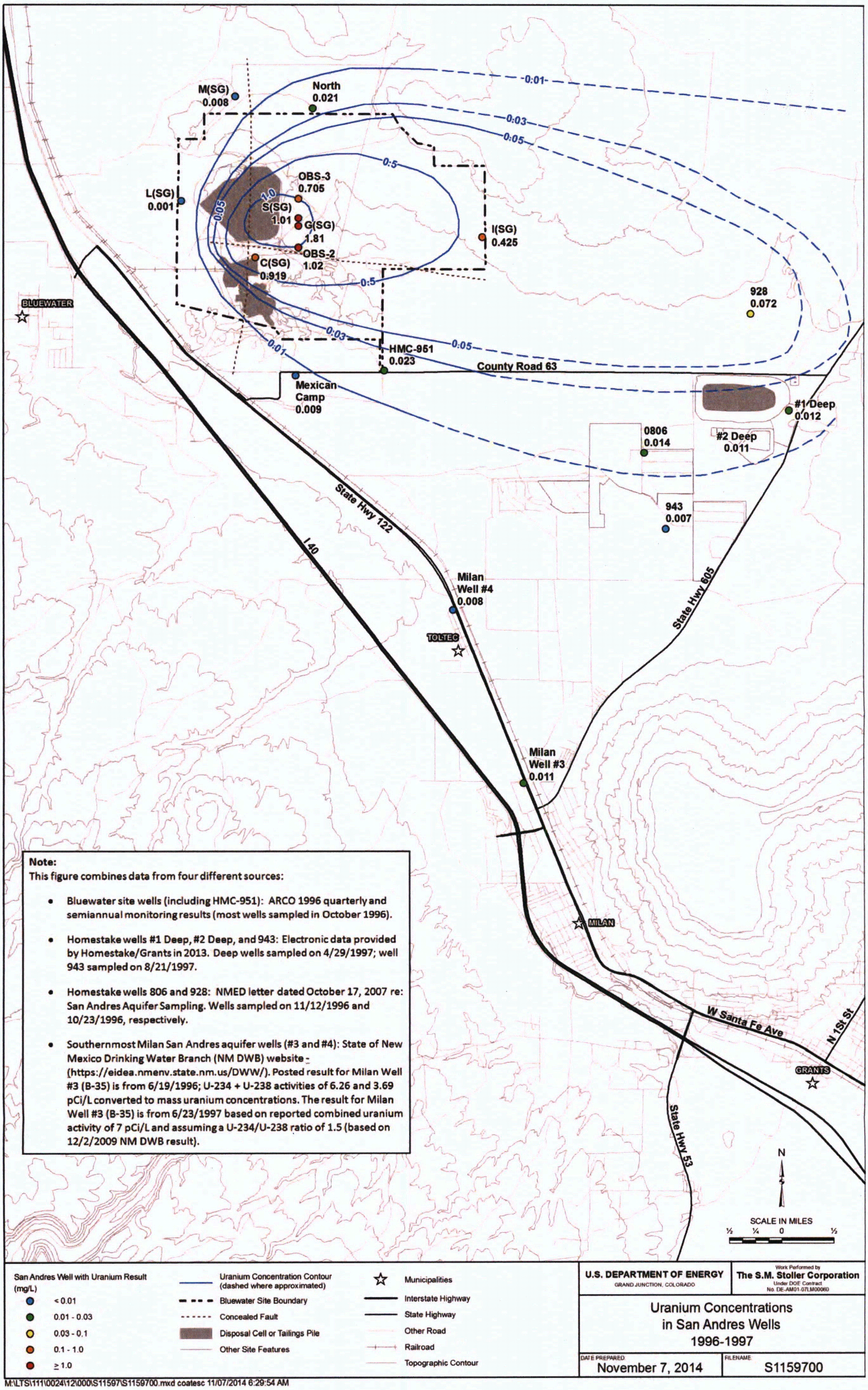
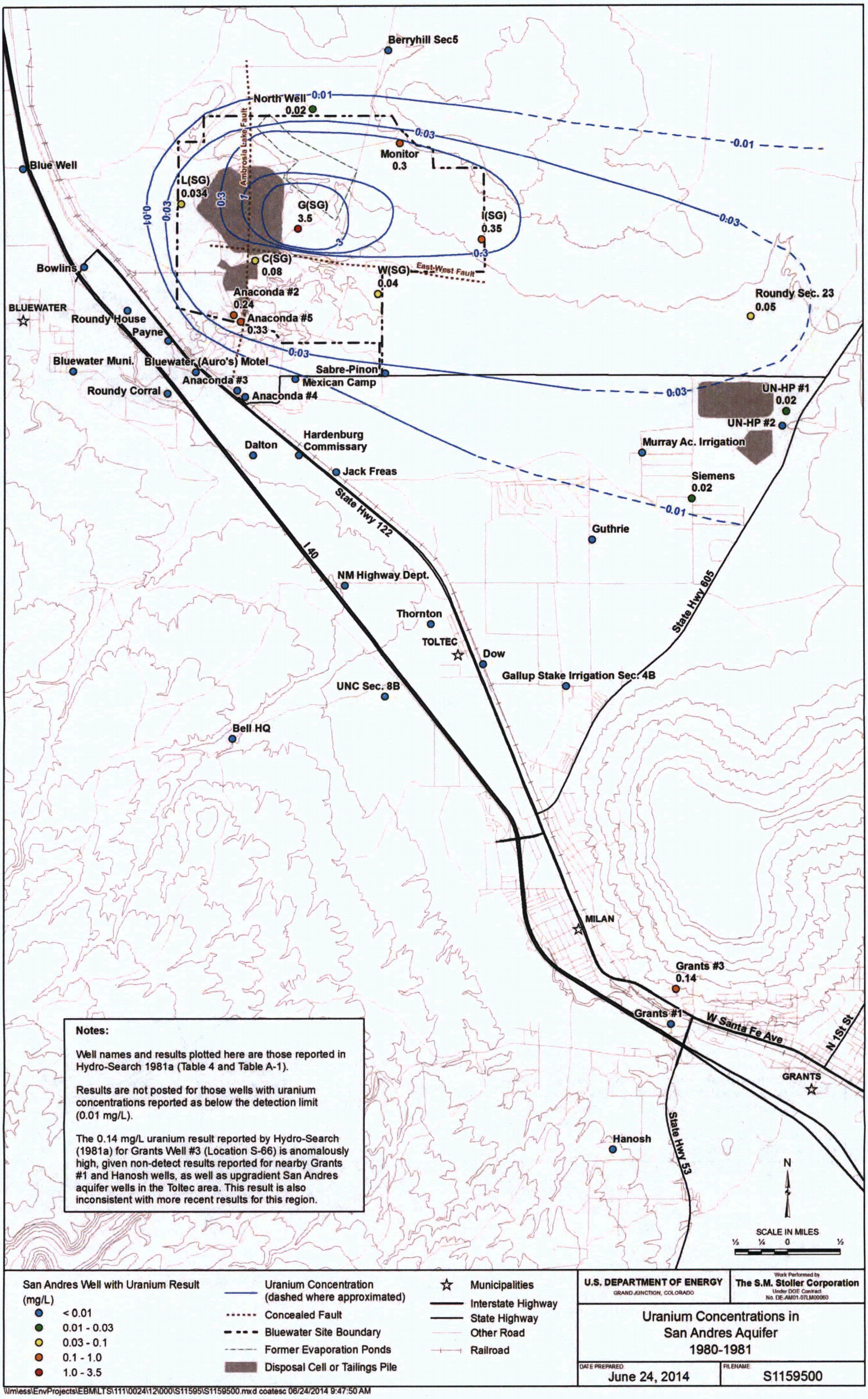
Figure 57. Regional Uranium Concentrations in the San Andres Aquifer in 2008



Note:
This figure combines data from four different sources:

- Bluewater site wells (including HMC-951): ARCO 1996 quarterly and semiannual monitoring results (most wells sampled in October 1996).
- Homestake wells #1 Deep, #2 Deep, and 943: Electronic data provided by Homestake/Grants in 2013. Deep wells sampled on 4/29/1997; well 943 sampled on 8/21/1997.
- Homestake wells 806 and 928: NMED letter dated October 17, 2007 re: San Andres Aquifer Sampling. Wells sampled on 11/12/1996 and 10/23/1996, respectively.
- Southernmost Milan San Andres aquifer wells (#3 and #4): State of New Mexico Drinking Water Branch (NM DWB) website - (https://eidea.nmenv.state.nm.us/DWW/). Posted result for Milan Well #3 (B-35) is from 6/19/1996; U-234 + U-238 activities of 6.26 and 3.69 pCi/L converted to mass uranium concentrations. The result for Milan Well #3 (B-35) is from 6/23/1997 based on reported combined uranium activity of 7 pCi/L and assuming a U-234/U-238 ratio of 1.5 (based on 12/2/2009 NM DWB result).

Figure 56. Regional Uranium Concentrations in the San Andres Aquifer in 1996–1997



Notes:

Well names and results plotted here are those reported in Hydro-Search 1981a (Table 4 and Table A-1).

Results are not posted for those wells with uranium concentrations reported as below the detection limit (0.01 mg/L).

The 0.14 mg/L uranium result reported by Hydro-Search (1981a) for Grants Well #3 (Location S-66) is anomalously high, given non-detect results reported for nearby Grants #1 and Hanosh wells, as well as upgradient San Andres aquifer wells in the Toltec area. This result is also inconsistent with more recent results for this region.

Figure 55. Regional Uranium Concentrations in the San Andres Aquifer in 1980–1981

observed at this location in 1980–1981. Moreover, uranium concentrations ranging from 0.007 to 0.05 mg/L were detected at additional wells in the San Andres aquifer (Figure 56 through Figure 58) in the vicinity of the Homestake site, supporting the possibility that dissolved uranium from the Bluewater site had migrated as far as the Homestake site as of 1980. With this possibility, it could be hypothesized that uranium concentrations measured at Homestake site wells screened in the San Andres aquifer after the mid-1990s were representative of steady-state concentrations that form upgradient of a contaminant plume's leading edge. Such a hypothesis implies that the leading edge of the Bluewater site uranium plume, as defined by a concentration of 1 mg/L, has been, for many years, east and east-southeast of the Homestake site, in the direction of areas north of Grants.

In contrast to the above-given finding that the dissolved uranium originating in San Andres aquifer groundwater at the Bluewater site migrated to the Homestake site as early as 1980, alternative arguments have occasionally been made to explain the presence of slightly elevated uranium levels in the aquifer at the Homestake site. In recent years, EPA has posited that uranium at bedrock aquifer wells in the vicinity of the Homestake site are the result of unintended effects of the groundwater remedy at the GRP. EPA (2011) implies that HMC has created a steep downward gradient between local San Mateo Creek alluvium and the underlying San Andres aquifer, which causes uranium contamination to migrate downward to the deeper aquifer along fault zones present at the Homestake site.

As with the alluvial aquifer, preparation of the uranium plume maps in Figure 55 through Figure 58 revealed the importance of including offsite data in this study. This exercise also suggests that it may be helpful to consider information from nearby sites when developing conceptual models for other LM sites. Without such additional information, the effects of historical contaminant processes at sites may go unnoticed.

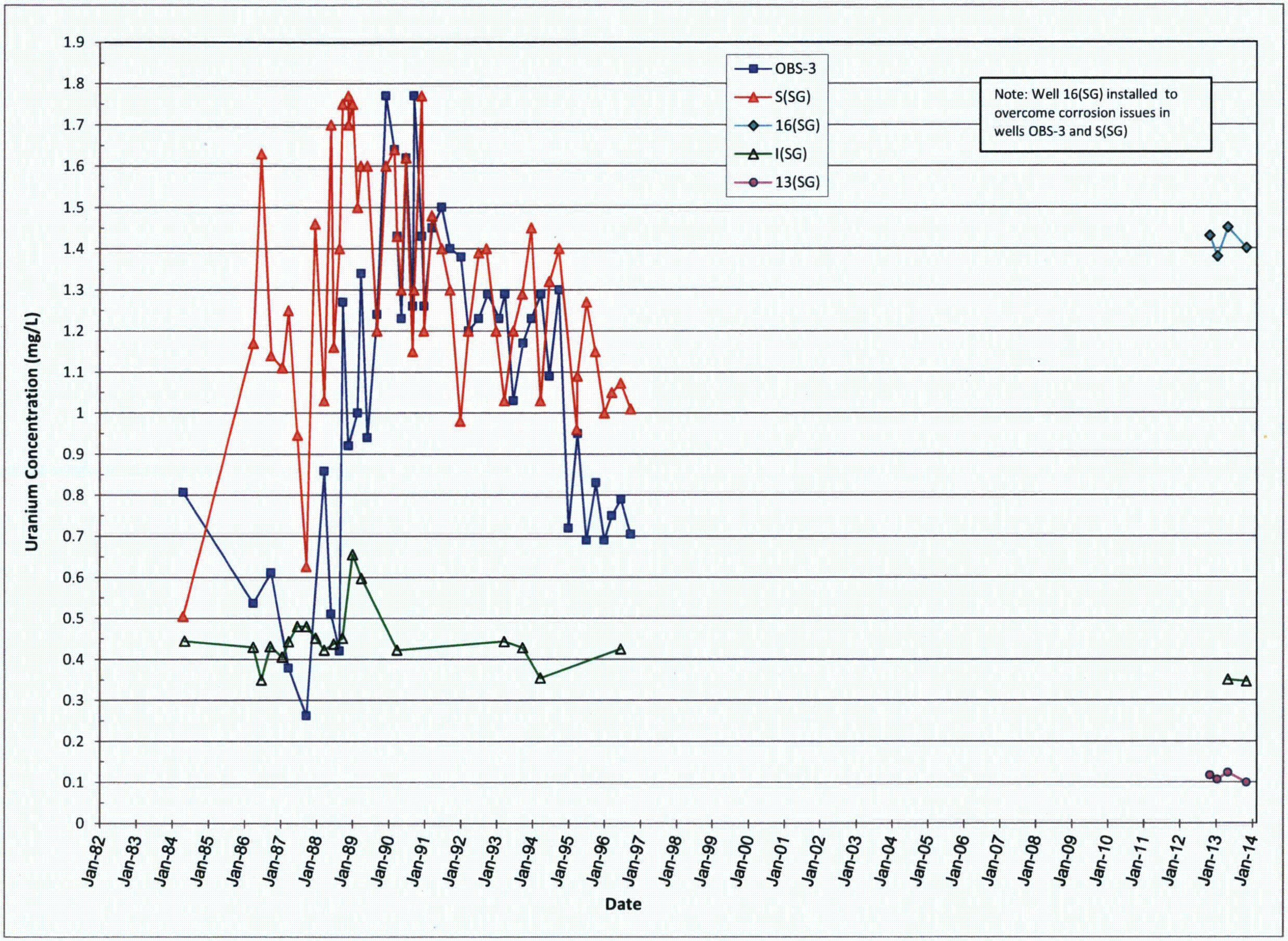
8.3.2 Uranium Concentration Histories

Prior to development of multiyear, temporal plots of uranium concentration at wells in the study area, the quality of the concentration data was reviewed to identify measured uranium concentrations that might be considered erroneous or not fully representative of the San Andres aquifer. The review indicated that uranium concentrations from Bluewater site wells OBS-3 and S(SG) in samples collected after 1996 were likely to misrepresent uranium contamination in areas just east of the main tailings disposal cell. Specifically, in 1997, samples from the two wells began to yield uranium concentrations that were anomalously low in comparison to concentrations from samples collected over the previous 13 years. This was not known at the time, however, because DOE did not have possession of historical monitoring data collected by ARCO. Because of this, the lower post-1996 concentrations at the two wells were considered signs that uranium contamination in the San Andres aquifer was attenuating in areas downgradient of the main tailings disposal cell. However, in-well video logs of the two wells in 2011 revealed that well screens in the two wells had been severely compromised by corrosion products. Subsequent data collected from the two wells indicated that the corrosion products were likely adsorbing much of the aqueous-phase uranium that had entered the casings of the wells. This observation, combined with the fact that low-flow sampling techniques had largely been used to monitor uranium in the wells since 2004, indicated that measured uranium concentrations at the two locations between 1997 and 2011 could not be trusted to represent contaminant levels in the surrounding aquifer. The effects of the well corrosion on uranium concentrations in samples collected from wells S(SG) and OBS-3 and corresponding uranium isotope activity ratios (ARs) were discussed in Section 7.2.3.

In response to the discovery of groundwater sampling issues stemming from corrosion in wells OBS-3 and S(SG), new well 16(SG) was installed in the San Andres aquifer just east of the main tailings disposal cell in summer 2012. Uranium concentrations measured in water samples collected from the new well since its installation have generally fallen in the range of concentrations seen at wells OBS-3 and S(SG) during the early 1990s (1.2 to 1.7 mg/L), suggesting that the aquifer east of the main tailings disposal cell continues to be impacted by steady mass loading of uranium from a mineralized zone beneath the cell.

Concentrations of uranium and other groundwater constituents monitored at well I(SG) have also been called into question. Well I(SG) has solid steel casing to a depth within the upper portion of the San Andres Limestone and then is uncased open borehole through the remainder of the San Andres Limestone and the Glorieta Sandstone. ARCO last sampled the well in 1996, using a high-flow casing purge method, and observed a uranium concentration of 0.42 mg/L; this result was consistent with results since 1984. DOE did not begin monitoring the well until 2008 because monitoring the well was not required by the LTSP; monitoring was initiated to support NMED's regional groundwater investigation. DOE collected samples from within the casing just above the open borehole using the low-flow method. Low uranium levels in DOE's samples were thought to be caused by natural attenuation processes in the San Andres aquifer. However, specific conductivities and uranium concentrations measured over the full water column in the well in 2013 revealed that contaminant levels are stratified within the aquifer, with higher conductivities and uranium concentrations observed at depth in the well. This finding indicated that the use of low-flow sampling techniques to collect water from the cased portion of the well resulted in uranium concentrations that were distinctly lower than those within the uncased portion. Sampling at the well since then, using techniques designed to capture water representative of groundwater in the limestone and sandstone surrounding the well, has yielded uranium concentrations only slightly lower than results observed by ARCO.

Upon completing the above-discussed review of sampling challenges at San Andres aquifer wells on the Bluewater site, a temporal plot of historical uranium concentrations at wells OBS-3, S(SG), 16(SG), I(SG), and 13(SG) was prepared (Figure 59). Because of the corrosion problems at wells OBS-3 and S(SG), and contaminant stratification issues in well I(SG), concentration data from 1997 through 2012 at these wells were omitted from the graph. Despite this shortcoming, the available data do indicate that uranium concentrations in onsite San Andres aquifer wells are remaining relatively stable. This finding is supported by measured uranium concentrations at well 16(SG) (1.3 to 1.5 mg/L), the well that is now used to monitor contaminant concentrations just east of the main tailings disposal cell. In addition, measured concentrations at well I(SG) in 2013 (0.35 mg/L) are at the low end of the range of concentrations recorded for this well from 1984 to 1996 (0.35 to 0.65 mg/L). Though constituent concentration data have only been collected at well 13(SG) since late 2012, uranium at this well in the southeast corner of the site has maintained a relatively constant concentration of about 0.1 mg/L. Taken together, the data plotted in Figure 59 suggest that mass loading rates to the San Andres aquifer for uranium residing in the mineralized zone beneath the main tailings disposal cell have been relatively stable, and that relatively steady or slowly decreasing uranium concentrations at onsite wells are reflective of portions of the uranium plume upgradient of the plume's leading edge.



Note: Well 16(SG) installed to overcome corrosion issues in wells OBS-3 and S(SG)

Figure 59. Temporal Plot of Uranium Concentrations in San Andres Aquifer Wells at the Bluewater Site

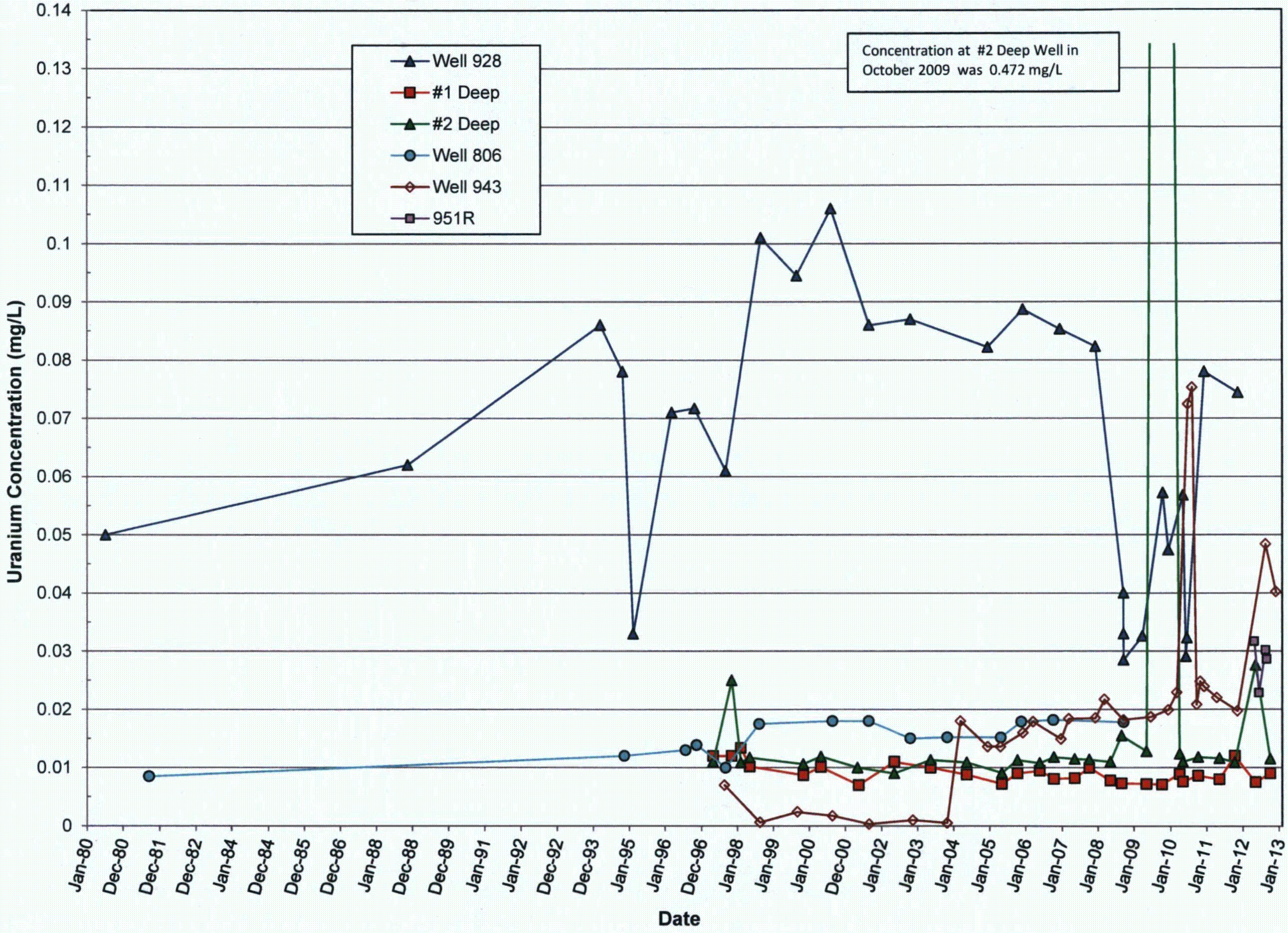


Figure 60. Temporal Plot of Uranium Concentrations in San Andres Aquifer Wells near the Homestake Site

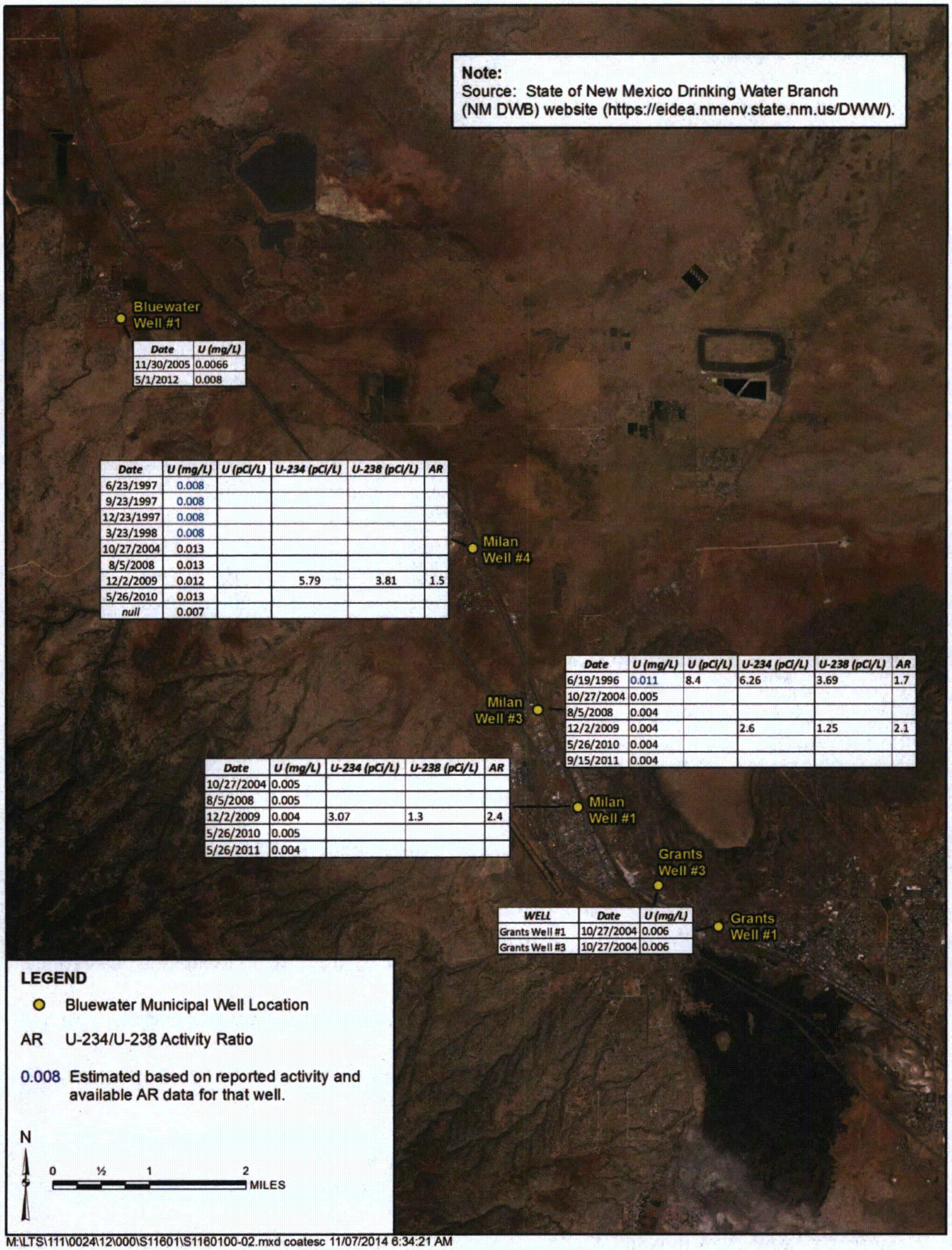
To assess the history of uranium concentrations at wells screened in the San Andres aquifer near the Homestake site, a temporal plot of concentrations at Homestake well locations 928, #1 Deep Well, #2 Deep Well, well 806, well 943, and well 951R was prepared (Figure 60). This graph suggests that the uranium concentration at well 928 has fluctuated between 0.035 and 0.105 mg/L since 1980. In contrast, concentrations in #1 Deep Well have been steady at about 0.01 mg/L, and uranium levels in well 806 have also remained steady, between about 0.01 and 0.02 mg/L. Other than an anomalously high concentration of 0.47 mg/L in 2009, uranium levels at #2 Deep Well have stayed within a range of about 0.01 to 0.03 mg/L (Figure 60). Uranium concentrations in well 943 have occasionally increased as high as 0.07 mg/L in recent years. Collectively, the historical concentrations could be considered relatively stable and representative of attenuated uranium contamination in portions of a plume that originates at the Bluewater site and extends some distance east of the Homestake site. With such a conceptualization, the leading edge of the uranium plume, as defined by a concentration of 0.01 mg/L is expected to be hydraulically downgradient of the Homestake site, in the direction of areas north of Grants.

The difficulties mentioned above concerning obstacles to acquiring contaminant concentrations representative of groundwater conditions in the San Andres aquifer serve as lessons that can potentially be applied to the monitoring of wells at other LM sites. Specifically, it important to examine all historical monitoring data for a site before LM takes responsibility for the long-term surveillance activities at the site. Periodic inspection of the variation in contaminant concentrations with depth in each monitoring well could also be beneficial, as would occasional video logs of the wells. It would also be helpful to compare the results of low-flow sampling with those from well purge sampling to ensure that the most representative concentration data are being collected. Finally, occasional critical assessments of the temporal concentration histories at individual wells, as discussed above, would help confirm the validity of data presented in annual monitoring reports.

8.3.3 Uranium Concentrations at Municipal Wells

Though the uranium plume maps shown in Figure 55 through Figure 58 indicate that uranium contamination in the San Andres aquifer has migrated eastward from the Bluewater site to the Homestake site, none of the maps imply that uranium has migrated west-southwest to Bluewater Village, or directly within the San Andres aquifer to Milan. However, Section 8.2.1 discussed the possibility that uranium-contaminated alluvial groundwater had migrated to an area near Toltec and was subsequently transported downward to parts of the aquifer tapped by Milan Well #4. To assess whether uranium contamination has affected groundwater withdrawn by the municipal wells, this study examined the full suite of historical uranium concentrations measured at drinking-water supply wells in the Grants-Bluewater Valley. Figure 61 illustrates these data, as published in databases maintained by the New Mexico Drinking Water Bureau.

As Figure 61 shows, uranium concentration data are available for community water-supply wells from various samples collected between 1997 and 2011. None of the posted concentrations exceed the uranium MCL of 0.03 mg/L, and most of the measured concentrations are less than 0.01 mg/L. The few cases in which the uranium concentration exceeds 0.01 mg/L are for samples collected from Milan Well B-50 (Milan Well #4) and Milan Well B-35 (Milan Well #3) during the 1990s. In general, the results shown in Figure 61 suggest that uranium contamination has not



Note:
Source: State of New Mexico Drinking Water Branch (NM DWB) website (https://eidea.nmenv.state.nm.us/DWW/).

Date	U (mg/L)
11/30/2005	0.0066
5/1/2012	0.008

Date	U (mg/L)	U (pCi/L)	U-234 (pCi/L)	U-238 (pCi/L)	AR
6/23/1997	0.008				
9/23/1997	0.008				
12/23/1997	0.008				
3/23/1998	0.008				
10/27/2004	0.013				
8/5/2008	0.013				
12/2/2009	0.012		5.79	3.81	1.5
5/26/2010	0.013				
null	0.007				

Date	U (mg/L)	U (pCi/L)	U-234 (pCi/L)	U-238 (pCi/L)	AR
6/19/1996	0.011	8.4	6.26	3.69	1.7
10/27/2004	0.005				
8/5/2008	0.004				
12/2/2009	0.004		2.6	1.25	2.1
5/26/2010	0.004				
9/15/2011	0.004				

Date	U (mg/L)	U-234 (pCi/L)	U-238 (pCi/L)	AR
10/27/2004	0.005			
8/5/2008	0.005			
12/2/2009	0.004	3.07	1.3	2.4
5/26/2010	0.005			
5/26/2011	0.004			

WELL	Date	U (mg/L)
Grants Well #1	10/27/2004	0.006
Grants Well #3	10/27/2004	0.006

LEGEND

●	Bluewater Municipal Well Location
AR	U-234/U-238 Activity Ratio
0.008	Estimated based on reported activity and available AR data for that well.

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Figure 61. Uranium Concentration Data for Municipal Wells in the Grants-Bluewater Valley

detrimentally affected municipal wells in the Grants-Bluewater Valley such that concentrations approaching the MCL of 0.03 are observed. However, it cannot be firmly concluded that uranium contamination originating in the alluvial aquifer has never reached groundwater that is pumped by one or more Milan municipal wells.

Figure 61 also lists the activity concentrations for the isotopes U-234 and U-238 at Milan Well #1, Milan Well #3, and Milan Well #4, which were addressed in Section 7.2.3. The corresponding AR values, also listed in Figure 61, are all greater than or equal to 1.5. The combination of these AR values with the full suite of uranium concentrations posted on the figure suggest that, if tailings-related uranium contamination reached any of the municipal wells at any time in the past, it was diluted by mixing with uncontaminated groundwater.

8.4 Uranium Transport Processes

The analyses and findings in this and preceding chapters suggest that multiple transport processes have affected uranium contamination in the alluvial and San Andres aquifers, as summarized in the following sections. Much of the following summary builds upon information presented in Appendix F, which comprises a general description of advective-dispersive transport phenomena in the subsurface. This appendix and the glossary in Appendix E provide a better understanding of the fate of uranium contamination originating in groundwater at the Bluewater site.

8.4.1 Advection

High groundwater velocities within secondary permeability features (fractures, solution channels, cavities) in the San Andres aquifer have historically facilitated and continue to promote rapid advection of uranium contamination originating at the Bluewater site (Section 4.3.3.2). Advective transport caused the uranium plume in the aquifer to migrate beyond the site's east boundary in less than 10 years. Moreover, this study concludes that the leading portion of the uranium plume had migrated as far as the Homestake site by 1980, more than 30 years ago. It is also likely that the plume in 1980 had migrated beyond the Homestake site, to the east-southeast in the direction of areas north of Grants (Section 8.3.1).

Though the average linear velocities associated with porous media flow of groundwater in the alluvial aquifer were generally lower than those attributed to secondary permeability features in the San Andres aquifer, they were nevertheless high enough to rapidly convey Bluewater site uranium contamination to offsite areas located southeast of the main tailings disposal cell. Consequently, the estimated travel times for alluvial aquifer groundwater between the site and downgradient areas of interest are relatively short. The travel-time estimates between the main tailings disposal cell and Toltec are generally on the order of 20 to 60 years (Section 4.3.3.2), providing sufficient time for uranium contamination mobilized in the alluvial aquifer in the mid-1950s to reach areas near Milan in recent years.

8.4.2 Dispersion

Relatively stable or slowly-decreasing uranium concentrations observed at onsite monitoring wells (Sections 8.2.2 and 8.3.2) suggest that a relatively continuous contaminant source at the Bluewater site is feeding uranium to plumes in the alluvial and San Andres aquifers. The mostly stable uranium concentrations in the plumes attenuate with flow distance between the main tailings disposal cell and the east boundary of the site (Sections 8.2.1 and 8.3.1).

As discussed in Chapter 7, transverse dispersion has historically been considered the primary mechanism for attenuating onsite uranium concentrations in the San Andres aquifer with flow distance, and for the relatively stable concentrations observed in site wells (Section 7.1.1.3). The analyses conducted in this study also indicate that transverse dispersion is a significant contributor, if not the most important contributor, to contaminant attenuation with flow distance, and is the primary cause of the uranium plume stability. Though most of this dispersion is assumed to occur in areas lateral to the centerline of the plume in the San Andres aquifer, transverse vertical dispersion is also likely contributing to the uranium attenuation.

A uranium plume influenced by mechanical dispersion is unlikely to maintain a perfectly symmetrical shape. Rather, an irregularly shaped plume is considered more likely in the San Andres aquifer because of groundwater flow through fractures, solution-enlarged joints, solution channels, and cavities. As a consequence, non-contaminated zones within the so-called plume boundaries are possible. It is also possible that the uranium plume in the San Andres aquifer bifurcates in some locations along preferential flow paths, such that uranium migration takes place in relatively thin, isolated plumes as opposed to a single, spatially continuous and wide plume. If this is the case, the capacity of transverse dispersion to attenuate plume concentrations would be enhanced, given that there is more plume surface area along which dispersion can take place with a bifurcated plume than is available with a single connected plume.

Mechanical dispersion of contaminants in site- and regional-scale plumes is generally attributed to aquifer heterogeneity (Domenico and Schwartz 1998). However, it has also been shown that transient flow effects in aquifers can lead to a form of apparent dispersion. This is because transient flow phenomena often cause changes in ambient flow direction over time, which in turn increases lateral spreading of a plume (e.g., Goode and Konikow 2000, Cirpka and Attinger 2003) beyond the spreading that would occur without such transient effects. Though difficult to discern given limited information on uranium transport in the vicinity of the Bluewater site, it is possible that apparent dispersion has also been promoted by decreasing groundwater elevations in the San Andres aquifer over the past 15 years. This is particularly true given that major changes in regional groundwater levels can alter flow directions in fractured-bedrock and karst aquifers (e.g., Huntoon 1995). With this perspective, temporally variable hydraulic heads in the region have the potential to enhance attenuation of the uranium plume in the San Andres aquifer in areas east and east-southeast from the Bluewater site.

Transverse dispersion is potentially increased in fractures and solution channels in the San Andres Limestone because these secondary permeability features act as conduits that are capable of conveying water at very high velocities (see Section 4.3.3.2). If the velocities increase to the extent that turbulent flow, as opposed to laminar flow, takes place, the degree to which contaminated groundwater mixes with fresh water is enhanced. Though it is unlikely that turbulent flow occurs throughout most of the karst-limestone portions of the San Andres Limestone, turbulent flow could occur in areas of rapid, convergent groundwater flow surrounding large-production pumping wells in the San Andres aquifer, such as the San Andres aquifer wells used to support groundwater remediation at the GRP.

Transverse dispersion has also influenced, and likely continues to influence, contaminant transport in the alluvial aquifer at the Bluewater site. The observed decrease in uranium concentration in ancestral river alluvium with distance from the contaminant source is consistent with advective transport in a downstream direction that is modified by transverse dispersion. As in the case of the San Andres aquifer, transient flow phenomena in the alluvial aquifer have the potential to alter groundwater flow directions, thereby inducing apparent transverse dispersion and enhanced attenuation of uranium concentrations.

8.4.3 Plume Stability

The development of a uranium plume in the San Andres aquifer fed by a continuous source and characterized by relatively stable concentrations due solely to the effects of transverse dispersion may seem counterintuitive. However, as discussed in Appendix F and textbooks on contaminant transport (e.g., Domenico and Schwartz 1998), virtually steady conditions can result from transverse dispersion alone. As a plume evolves, transverse spreading of contaminant mass causes concentrations to progressively decrease with flow distance, eventually leading to contaminant levels at the plume front that are indistinguishable from the background concentration. In effect, enough time has elapsed and the plume has migrated sufficiently far downgradient that concentrations outside the zone of constant concentration (i.e., along the plume edges) are so low as to be considered inconsequential. At this later time, loss of contaminant mass along the plume's border, as defined by the background concentration, occurs at the same rate contaminant mass is being added to the aquifer from the plume source area.

Figure 62 illustrates conceptually how steady conditions gradually evolve in a plume that is supplied by a continuous source with concentration C_o and is affected by both longitudinal and transverse dispersion. This graph shows resulting concentration-versus-distance profiles along the centerline of the plume at six different times (t_1 through t_6), with each successive profile indicating that a larger portion of the plume has stabilized (i.e., reached a steady state). Steady concentrations first occur on the downstream edge of the source, and are observed upon the initial onset of contamination in the groundwater. As time progresses, the steady-state portion of the profile constantly expands in the downgradient direction. This process continues until time t_6, when all concentrations in the plume greater than the background concentration no longer change.

The graph in Figure 62 can be used to describe the evolution of the uranium plume in the San Andres aquifer that originates at the main tailings disposal cell on the Bluewater site and extends east-southeastward to the Homestake site and areas north of Grants. This is accomplished by assuming that location S_0 in the figure represents the east edge of the main tailings disposal cell, location S_s represents the east boundary of the Bluewater site, location S_h represents the Homestake site, and location S_b represents an area downgradient of the Homestake site. Under these assumptions, the concentration-versus-distance profiles indicate that steady-state concentrations are achieved between the plume source and the east boundary of the Bluewater site at time t_2, and the steady concentration at the site boundary is C_s. This in turn indicates that, at time t_2, the onsite portion of the plume has stabilized, and all plume areas downgradient of the site are in a transient state. Similarly, steady-state concentrations are observed in all areas of the plume upgradient of the Homestake site as of the later time t_4, with a stable concentration of C_h at this site. Figure 62 shows contaminant levels stabilizing at the background concentration C_b as of time t_5 at location S_b, which is downgradient of the Homestake site.

Current conditions within the uranium plume in the San Andres aquifer correlate with the concentration profile shown for time t_6. This curve indicates that concentrations downgradient of location S_b will remain less than the background concentration C_b, even though the plume front can continue to migrate downgradient of location S_b. This makes it difficult to distinguish uranium contamination, if any, in areas north of Grants from naturally derived uranium in the aquifer.

A background concentration was used here to demonstrate the progression of steady-state conditions during plume evolution in the San Andres aquifer because uranium levels above the regional background concentration (0.01 mg/L) are of greatest interest to this investigation. However, an alternative concentration, such as the New Mexico drinking water standard for uranium (0.030 mg/L) could just as easily be used to delineate the steady plume border. The conceptual model described in this report implies that relatively stable concentrations equal to the standard occur somewhere east of the Bluewater site and perhaps in the vicinity of the Homestake site.

The discussion above regarding plume stability is strictly theoretical in the sense that steady-state concentrations are rarely, if ever, observed in groundwater plumes. In real groundwater systems, fluctuations of measured concentration at each location in space are a natural consequence of hydrologic and transport processes and measurement error (see, for example, Figures 55, 60, and 61). Nonetheless, the concentrations at each point within a so-called stable plume tend to fluctuate around an average, representative value for that point, instead of showing an increasing or decreasing trend. These average concentrations are, in effect, representative of the steady concentrations that would be observed in a theoretical system.

The concentration profiles in Figure 62 for conditions along the plume centerline can be generated using a transient model of advective-dispersive transport that accounts for one-dimensional advection and three-dimensional dispersion. Any of several numerical models, such as those based on finite-difference or finite-element techniques, could be employed for this purpose, eventually producing an effectively stable plume. By using a more simplified approach, the plume evolution could be approximated with an analytical mathematical solution to the governing transport equation. Examples of applicable analytical models include those described in Leij and Bradford (1994), Falta et al. (2007), and Karanovic et al. (2007). Domenico (1987) used an analytical solution to the transient advection-dispersion equation to illustrate that the steady concentrations produced solely by transverse dispersion occur in areas some distance upgradient of the plume's advective front, which is defined as the product of average linear velocity and the time since the onset of contamination in the groundwater. The distance separating the downgradient extent of steady concentrations from the advective front is small in cases where the influence of longitudinal dispersion is relatively minor in comparison to the influence of advection (Domenico 1987).

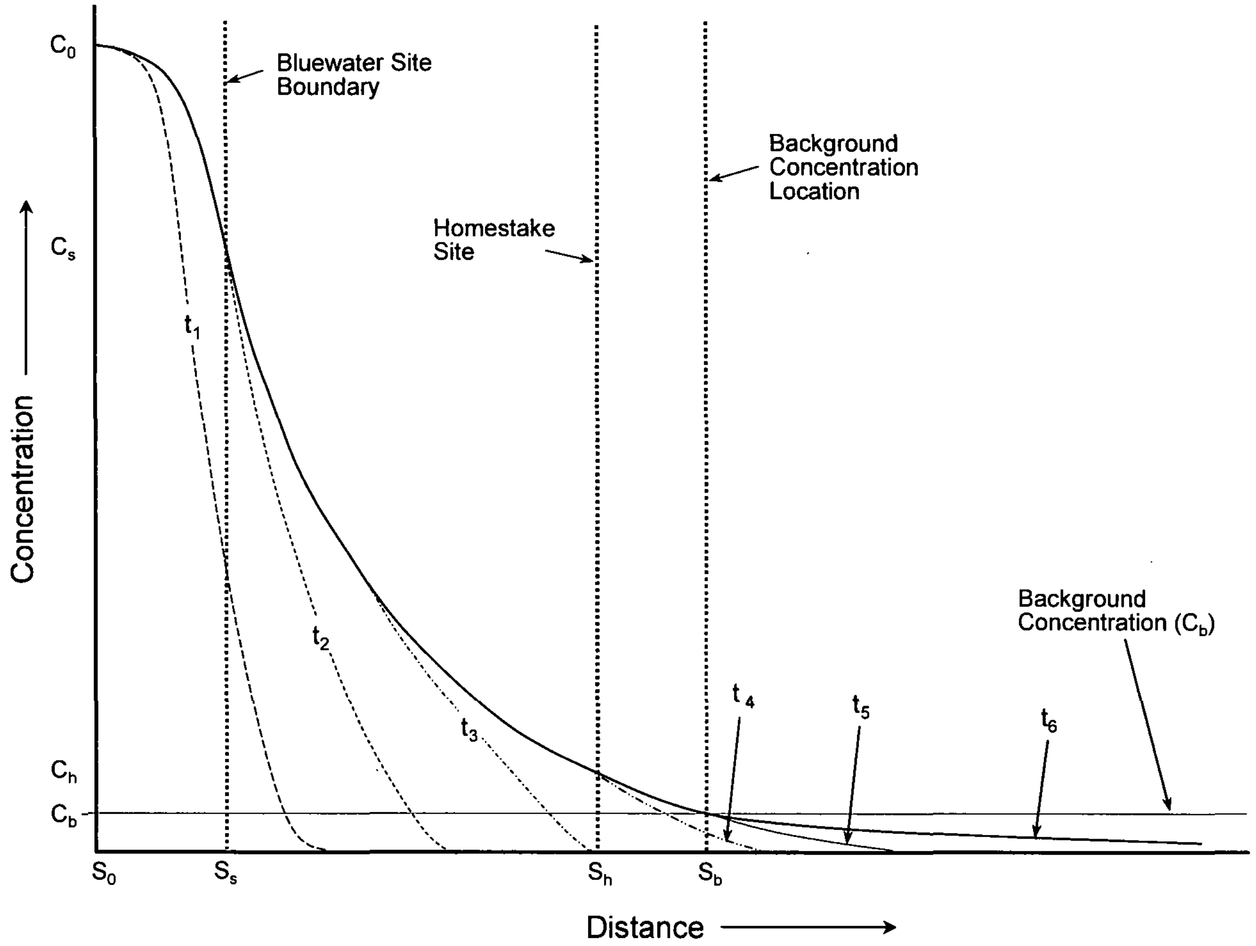

Figure 62. Graphical Depiction of Contaminant Plume Evolution—Concentration Profiles Along the Plume Centerline at Successive Times t_1 Through t_6

For groundwater systems dominated by advective transport in the direction of flow and contaminant spreading normal to the flow direction, the concentrations in a stable plume caused solely by transverse mixing can be calculated directly using analytical solutions to the steady-state version of the transport equation (e.g., Domenico and Palciauskus 1982, Domenico and Robbins 1985, Leij and Bradford 1994). The mathematical derivations under this category assume that transverse concentration gradients determine the ultimate width of the plume and that longitudinal dispersion is an insignificant process. In some cases, this assumption is made on the basis of experimental findings by Harleman and Rumer (1963) during a laboratory investigation of the relative effects of longitudinal and transverse dispersion. An analytical steady-state model could be used to estimate the downgradient extent of the stable uranium plume that originates at the Bluewater site and extends east-southeast of the Homestake site.

8.4.4 Merged Plumes

Concentrations of uranium in the alluvial aquifer plume originating at the Bluewater site appear to decrease with flow distance in areas immediately southeast of the site's east boundary. However, the uranium concentrations in the plume start to increase about a mile southeast of the Bluewater site property. This is attributed to the merging of the Bluewater plume with a separate plume migrating westward from the Homestake site (Section 8.2.1). Though the relative contributions of the individual plumes to the combined plume have not been quantified, both the volumetric flow rate and the uranium concentrations associated with the Homestake-related plume appear, in recent years, to be larger than the flow and concentrations associated with the Bluewater site plume. Thus, the apparent increase in uranium concentrations about a mile southeast of the Bluewater site's east border can be attributed to mass discharge of uranium from the Homestake site. Continued groundwater remediation at the GRP has the potential to reduce the Homestake site's contribution to the merged plume in the alluvial aquifer.

8.4.5 Geochemical Reactions

The chemical data used to characterize the oxidation state of the alluvial and San Andres aquifers indicate that mostly oxidized conditions are prevalent (Sections 7.2.1.1 and Sections 7.2.1.2). Under such conditions, most of the uranium migrating in the aquifers is expected to occur as highly mobile uranyl species (Section 7.2.2.3). Assessment of whether uranium in groundwater at the Bluewater site could precipitate in mineral form indicates that all uranium minerals are undersaturated by at least 3 orders of magnitude (Section 7.2.2.2). Thus, it is unlikely that uranium minerals are chemically precipitating in the plumes originating at the Bluewater site.

8.4.6 Uranium Adsorption

Adsorption of uranium on mineral surfaces within the alluvial and San Andres aquifers is potentially an effective mechanism for retarding the migration of the uranium plumes originating at the Bluewater site. Though hydrous ferric oxide is expected to be present in the alluvial and San Andres aquifers, limited information regarding its abundance makes it difficult to predict how adsorption of uranium will impact the Bluewater site plumes (Section 7.2.2.1).

8.4.7 Dual-Domain Effects

Groundwater migrating through limestone and fractured sandstone in the San Andres aquifer is subject to contaminant transport processes associated with a coupled fracture-matrix flow system (Zimmerman et al. 1993). The dual-domain transport occurring in such a flow system acts to slow, or retard, the migration of uranium to downgradient areas, in a manner similar to the plume retardation caused by uranium adsorption. If the mechanisms by which uranium contamination was released from the Bluewater site constituted a finite-mass, pulse source, dual-domain transport could provide an effective means of attenuating the uranium plume in the aquifer during future years (Appendix F). However, this type of attenuation is unlikely to take place, given that uranium appears to be released to the San Andres aquifer by a continuous, constant-concentration source. As a result, uranium contamination is expected to persist in the aquifer for at least decades and perhaps hundreds of years.

Dual-domain transport processes also likely affect uranium contamination in the alluvial aquifer. The coarse-grained sands and gravels in paleochannel portions of ancestral Rio San Jose alluvium act as preferential pathways for mobile-phase contamination, whereas finer-grained deposits in non-channel parts of the alluvium retain and slowly release relatively immobile contamination to the preferential pathways. As with the San Andres aquifer, the heterogeneous fluvial deposits in the alluvial aquifer will cause uranium contamination in the aquifer to persist for many years.

8.5 Projected Fate of Uranium Contamination

The analyses presented in this and preceding chapters suggest that dissolved uranium contamination originating at the Bluewater site will persist indefinitely in the alluvial and San Andres aquifers. Moreover, uranium concentrations in groundwater in areas immediately downgradient of contaminant sources at the site are expected to either remain constant or decrease very slowly (Sections 8.2.2 and 8.3.2) in coming years.

Natural attenuation processes have apparently caused uranium concentrations in the San Andres aquifer in the vicinity of the Homestake site to remain relatively low since 1980, though some concentrations exceeding the uranium MCL (0.03 mg/L) have been observed in San Andres aquifer wells at the Homestake site since that time (Section 8.3.2). Given the history of uranium measurements at San Andres aquifer wells at the Homestake site and the apparently stable nature of uranium contamination in the aquifer at the Bluewater site (Section 8.4.2), relatively constant uranium concentrations similar to those measured between 1980 and 2013 (Section 8.3.2) at Homestake site wells are expected for the indefinite future. Large, permanent increases or decreases in uranium concentration are not expected.

The conceptual model in this study suggests that uranium concentrations in the San Andres aquifer downgradient of the Homestake site, in areas directly north of Grants, are less than the assumed background uranium concentration of 0.01 mg/L. Because the conceptual model indicates that uranium concentrations in the vicinity of the Homestake site will remain relatively constant, uranium concentrations above the background level are not expected in the areas directly north of Grants.

8.6 Potential Effects of Increasing Regional Groundwater Levels

It is possible that uranium concentrations in the aquifers will show temporary increases if and when regional groundwater levels begin to recover from the observed decreases in hydraulic head seen over the past 15 years (Section 4.3.5). At the Bluewater site, such increases in concentration could be caused by the upward incursion of groundwater into, and enhanced mobilization of uranium from, parts of the mineralized zone that have been dominated by unsaturated conditions during the past several years. The magnitude and duration of the temporary increases in uranium concentration, if they do occur, are difficult impossible to predict.

8.7 Potential for Source Removal

Because dissolved uranium in the alluvial and San Andres aquifers at the Bluewater site is sourced by dissolution of solid-phase uranium in a thick mineralized zone beneath the main tailings disposal cell and to a lesser extent the carbonate tailings disposal cell, removal of the contaminant source is considered unfeasible. The mineralized zone likely extends more than 100 ft deep below the base elevation of the cells and laterally along fault zones.

Assessing the extent of the mineralized zone would require numerous characterization boreholes beneath the disposal cells and in areas downgradient of the cells. Physical methods of source removal would likely require relocation of the cells and would require considerable excavation to remove the mineralized materials. Dewatering the deeper portions of the excavation to allow material extraction from the contaminated San Andres Limestone and Glorieta Sandstone may be impossible because of the hydraulic characteristics of the San Andres aquifer.

Source removal through hydraulic means, such as groundwater pumping, is also expected to be unsuccessful. The groundwater withdrawn by pumping would mostly migrate through preferential flow paths that do not come in direct contact with much of the solid-phase contamination in the mineralized zone beneath the main tailings disposal cell. After periods of non-pumping, contaminant rebound due to back diffusion of contamination from the less permeable portions of the mineralized zone would likely occur.

9.0 Groundwater Conceptual Model

The primary purpose of this study was to develop a groundwater conceptual model that describes the aquifers associated with the Bluewater site and the potential exposure of downgradient groundwater users to mill-related contamination. A study area encompassing approximately 195 square miles was used to incorporate the contaminant source areas, the hydrogeologic features that most directly influence groundwater flow and contaminant transport in the Grants-Bluewater Valley, and the points of groundwater use, particularly the communities of Milan and Grants.

The conceptual model is based on work performed to meet the study objectives listed in Section 1.3 of this report. The activities involved in developing the model revealed numerous uncertainties that affect interpretations of groundwater flow and contaminant transport in the study area, which are addressed in Chapter 10. Despite such uncertainties, the following observations and conclusions have been drawn from this study.

9.1 Sources of Groundwater Contamination

The Anaconda/ARCO Bluewater uranium mill operations contaminated the two uppermost aquifers at the site: an aquifer in the ancestral Rio San Jose alluvial system, and the San Andres aquifer composed of the hydraulically connected San Andres Limestone and Glorieta Sandstone formations. Fluids from the mill's tailings impoundments seeped into the two aquifers, and contaminants in the fluids were apparently left in solid form within a mineralized zone in the geologic strata (basalt, alluvium, limestone, and sandstone) and fault zones under the impoundments (Section 8.1.1). This apparent mineralized zone is considered to be a continuing source of groundwater contamination in both aquifers.

Operations at the Homestake uranium mill contaminated the San Mateo Creek alluvial aquifer underlying the site's tailings piles as well as sandstone units of the underlying Chinle Formation (Section 2.4). Contamination at the Homestake site is affected by inflowing alluvial groundwater that is contaminated by processes occurring in areas north of the former mill site, including natural dissolution of uranium-bearing rock within the San Mateo Creek basin and decades of uranium mining and milling in a portion of the basin upgradient of the Homestake site. HMC is in the process of remediating site-derived contamination in the alluvial and affected Chinle aquifers.

9.2 Geological Features that Affect Groundwater Flow

Alluvium of the ancestral Rio San Jose at the Bluewater site is buried by multiple flows of Bluewater Basalt. The alluvial materials are in a former river valley that existed in the south portion of the site, including under a substantial portion of the main tailings disposal cell. The alluvial sediments within this paleochannel combine with San Mateo Creek alluvium about a mile southeast of the Bluewater site. Paleochannels in the two alluvial systems merged to form one channel that coursed southeastward in the direction of Milan. Alluvial groundwater flow paths follow these paleochannels (Section 4.2.1).

Two major faults intersect the Bluewater site under the south portion of the main tailings disposal cell and result in four fault blocks at the site. Movement along the faults and subsequent erosion caused the surface exposure of the San Andres Limestone under and adjacent to the southeast corner of the main tailings disposal cell. Vertical displacement along portions of the faults resulted in the faults acting as partial hydraulic barriers to horizontal groundwater flow at some site locations (Section 4.3.4). The fault zones are also apparently capable of acting as vertical and horizontal conduits of groundwater flow.

Two major faults that influenced the formation of the San Mateo Creek valley underlie the Homestake site. It is unclear whether the faults act as partial hydraulic barriers in the San Andres aquifer in the vicinity of the Homestake site. Neither the USGS (Baldwin and Anderholm 1992, Frenzel 1992, Gordon et al. 1960) nor HMC (e.g., HMC [2012] and HMC and Hydro-Engineering [2014]) discuss the potential influence of the faults on regional groundwater flow in the aquifer.

The geology of the San Andres Limestone and Glorieta Sandstone provides an environment for a highly productive aquifer. Pumping rates from wells completed in karst features (i.e., solution-enlarged joints, channels, and cavities) in the upper portion of the San Andres Limestone can be 1,000 gpm or higher, and pumping usually creates minimal drawdown. Faults and fracture zones within both formations enhance the productivity of the aquifer and can also serve as conduits for localized high groundwater flow rates. Groundwater also flows through the unfractured, porous sandstone strata within the aquifer.

9.3 Groundwater Sources and Flow

9.3.1 Alluvial Aquifer

The alluvial aquifer in the Grants-Bluewater Valley consists of three different types of alluvium that are distinguished by their respective fluvial sources: ancestral Rio San Jose alluvium underlying the Bluewater Basalt; recent Rio San Jose alluvium along the west and southwest edges of the Bluewater Basalt; and San Mateo Creek alluvium in the east end of the valley. The three alluvium types blend together in the vicinity of Milan. The alluvial aquifer downgradient of Milan is associated with the southeast-coursing Rio San Jose downstream from its confluence with San Mateo Creek. The point of confluence for the river and creek appear to have varied, both over geologic time and during the 19th and 20th centuries.

The ancestral Rio San Jose alluvium is recharged by precipitation on and subsequent downward seepage through the overlying Bluewater Basalt; subsurface inflow from the recent Rio San Jose alluvium in areas west and northwest of the Bluewater site; seepage losses from the current Rio San Jose channel; and infiltration of irrigation water. The ancestral alluvium was also recharged by seepage of tailings fluids during milling operations at the Bluewater site. Groundwater in the ancestral river system flows southeastward within channel sediments buried beneath the south half of the site. This groundwater subsequently merges with westward-flowing groundwater in the San Mateo Creek alluvium, beginning about a mile southeast of the Bluewater site. The combined flows continue to the southeast toward Milan and Grants.

ARCO's hydrology consultants estimated transmissivity, hydraulic conductivity, and storativity values for the ancestral alluvium based on aquifer pumping tests conducted at the Bluewater site. The calculated hydraulic conductivity values are indicative of highly permeable sands and gravels deposited in paleochannels of the ancestral Rio San Jose. Flow patterns and velocities in the buried alluvium are assumed to be variable because of an apparent complex distribution of channel sediments and associated floodplain deposits. Nonetheless, alluvial aquifer water levels suggest that a main paleochannel traverses the south half of the site. This is assumed to be the river channel that existed when the first lava flow of the Bluewater Basalt filled the alluvial valley. Groundwater flow velocities (and associated contaminant transport by advection) would be highest in this channel (Section 4.2.1).

A variety of water sources recharge the recent Rio San Jose alluvium. Sources include seepage losses from perennial Bluewater Creek where it emerges from Bluewater Canyon in the Zuni Mountains; infiltration of precipitation on the alluvium; downward seepage of surface water flows in irrigation canals fed by Bluewater Creek; seepage from the current channel of the ephemeral river; seepage of surface water diverted from the river into a borrow pit about a half-mile south of the Bluewater site; infiltration of applied irrigation water; and occasional seepage losses from tributary drainages at their outlets at the base of the Zuni Mountains. Groundwater flow directions in the recent alluvium vary from northeastward to eastward and southeastward depending on location. The recent river alluvium is not impacted by contamination originating at the Bluewater site.

San Mateo Creek alluvium in the Grants-Bluewater Valley is recharged by precipitation within the drainage basin, infiltration of applied irrigation water in areas south and west of the Homestake site, and subsurface alluvial inflow from upgradient areas. Historically, much of the subsurface inflow was derived from mining and milling operations (e.g., mine dewatering, seepage from tailings) in upgradient portions of the basin, primarily in the Ambrosia Lake area. Recharge also occurred as tailings fluids seeped from the Homestake tailings piles. Groundwater in San Mateo Creek alluvium generally flows southward toward Milan, but a component of flow is directed west from the former mill area within a paleochannel that terminates at the paleochannel containing ancestral river alluvium and extending southeastward from the Bluewater site.

9.3.2 San Andres Aquifer

The San Andres aquifer is recharged from Bluewater Lake and Bluewater Creek west of the Bluewater site and from precipitation on exposed outcrops of the San Andres Limestone and Glorieta Sandstone on the north slope of the Zuni Mountains. Groundwater flow in the aquifer beneath the site is in the east-southeast direction. Groundwater flows from the Bluewater site toward the Homestake site, and then continues in an east-southeast to southeast direction toward areas north of Grants (Section 4.3.3.1).

Prior to the start of large-scale groundwater extraction activity in the 1940s, groundwater elevations (hydraulic heads) in the aquifer in the Milan/Grants region were high enough to cause upward flow into overlying alluvial deposits as well as discharge to a large spring (Ojo del Gallo) on the southeast end of the Zuni Mountains. Since then the spring has ceased flowing, and the potentiometric surface for the aquifer in the Grants-Bluewater Valley has varied greatly in response to temporal variations in regional pumping and aquifer recharge in the Zuni Mountains.

Flow paths for the groundwater migrating in the San Andres aquifer from the Bluewater site do not intersect the locations of water supply wells for the municipalities of Bluewater and Milan (Section 4.3.3.1). The municipal well for the Village of Bluewater is upgradient of the Bluewater site, and recharge to that well is primarily from the west and northwest. South of the village, northeastward- and eastward-moving fresh groundwater derived from recharge on the north slopes of the Zuni Mountains pushes groundwater in the Grants-Bluewater Valley about 1.5 mi farther to the north than would occur if this type of recharge were not present. Consequently, San Andres aquifer flow paths from the Bluewater site pass well north of the Milan municipal wells.

Darcy's Law calculations suggest that average linear velocities of groundwater in the San Andres aquifer cover a wide range (Section 4.3.3.2). The highest velocities appear to be associated with conduit flow in the karst portions of the San Andres Limestone. Estimated velocities in secondary permeability features (fractures) in sandstones are somewhat lower than those in karst features, but they appear to be sufficiently high for conveying contaminants significant horizontal distances via advection. In contrast, estimated average linear velocities associated with porous media flow in unfractured sandstone are low, suggesting that contaminant migration in the sandstone is mostly vertical, occurring as either molecular diffusion or very slow advection away from or into the secondary permeability features consisting of karst solution channels or sandstone fractures.

As a result of the widely varying velocity estimates, horizontal flow velocities are expected to vary greatly over the vertical sequence of limestones and sandstones that compose the aquifer. Estimated groundwater travel times between the Bluewater main tailings disposal cell and the Homestake large tailings cell via conduit flow in the karst system, the uppermost and most used portion of the aquifer, vary from 2 years to 52 years, depending on hydraulic conductivity and effective porosity values used in the flow calculations (Table 5). Uranium concentrations in water samples collected from San Andres aquifer wells in the vicinity of the Homestake site suggest that groundwater impacted by milling activity at the Bluewater site had already reached the Homestake area as early as 1980, which is about 24 years since seepage from the main tailings impoundment began.

Groundwater withdrawals from the San Andres aquifer influence local flow directions and hydraulic gradients. Historical pumping of the Anaconda production wells, located south (cross-gradient) of the tailings impoundments at the Bluewater site, produced a cone of depression that drew contaminated groundwater toward the well field (Section 7.1.1.2). Historical withdrawals from the San Andres aquifer in the Milan/Grants area helped contribute to large decreases in hydraulic head that eventually led to the drying up of Ojo del Gallo spring.

Hydraulic head changes in the aquifer in the Toltec/Milan area appear to have caused a local reversal of the vertical gradient between the alluvial and San Andres aquifers (Section 8.2.1). Whereas upward flows from the bedrock aquifer to the alluvial aquifer occurred under natural conditions, the flow direction is mostly downward today due to groundwater pumping from the San Andres aquifer at the Milan municipal wells. Although undetermined at this time, it is possible that pumping from the San Andres aquifer in support of groundwater remediation at the Homestake site may have changed the vertical hydraulic gradient between the alluvial and San Andres aquifers from upward, under natural conditions, to downward under current conditions. EPA has suggested that such a gradient reversal may cause downward migration of contaminants from shallow alluvium to the bedrock aquifer along portions of the San Mateo fault system in the Homestake site area (EPA 2011).

9.4 Ambient Chemistry of Water Resources

Ambient, or background, water chemistry in the study area is important because it affects how contaminants dissolved in groundwater, particularly uranium, are transported from contaminant sources to downgradient monitoring wells.

9.4.1 Surface Water

Surface water features in the study area include Bluewater Lake, Bluewater Creek, Rio San Jose, and San Mateo Creek. Water quality information for these features is limited. Water quality data are available for Bluewater Lake and Bluewater Creek, the only perennial surface water sources on the north slope of the Zuni Mountains. These data are germane to this study because both the lake and the creek provide recharge to parts of the San Andres aquifer that flow beneath the Bluewater site. The Rio San Jose and San Mateo Creek in the Grants-Bluewater Valley only flow when high-intensity rainfall events produce sufficient runoff. Two perennial flow locations in the San Mateo Creek drainage several miles upgradient of the Homestake site (and upgradient of uranium mining and milling) have water quality data.

The dissolved oxygen and dissolved iron concentrations in streams in the area indicate that the water is oxidized, and any uranium in them would be in the U(VI) oxidation state (Section 5.1). The water is slightly basic and has relatively low to moderate salinity. All surface water samples are undersaturated with gypsum, so gypsum would be dissolved in these waters. Contaminant concentrations are unknown for surface water samples upgradient of the Bluewater site, but sample results suggest that surface watercourses in the upper San Mateo Creek drainage are not receiving high fluxes of arsenic, molybdenum, or selenium.

9.4.2 Alluvial Aquifer

Water quality data were evaluated from wells located in Rio San Jose alluvium upgradient of the Bluewater site and from wells completed in San Mateo Creek alluvium in two areas upgradient of the Homestake site. One of the alluvial aquifer wells in the San Mateo Creek drainage is directly north of the Homestake site, and the other is in the upper reaches of the drainage basin upgradient of mining and milling activity.

The chemical signatures of alluvial aquifer groundwater are indicative of the rock types or alluvial materials that the groundwater passes through. The available chemical data for the aquifer show differences between well locations, but most water samples collected from the wells are relatively oxidized, suggesting that precipitation of reduced uranium minerals does not occur (Section 5.2.1).

Samples from the background Rio San Jose alluvial aquifer wells are nearly equivalent in bicarbonate and sulfate, and have higher calcium and lower sodium equivalents than samples collected from wells in the San Mateo Creek alluvium. There was no uranium mining or milling upgradient of the Bluewater site. Correspondingly, no contamination is detected in the background wells for the Rio San Jose alluvial aquifer.

The alluvial groundwater with the highest salinity and highest concentrations of uranium and selenium is in the area directly north of the Homestake site. Groundwater in the alluvium in this area represents the dissolution product of gypsiferous and uranium-bearing formations in the San Mateo Creek drainage basin, as well as contamination from mining and milling in the basin. Consequently, the ambient quality of the San Mateo Creek alluvial groundwater entering the Homestake site is degraded by elevated contaminant concentrations, with uranium and selenium concentrations exceeding the drinking water standard in some of the alluvial aquifer wells (Section 5.2.1). A background uranium concentration for groundwater unaffected by mining and milling operations has not been determined for the San Mateo Creek alluvial aquifer.

Applied Hydrology Associates, Inc. (1990) reported that uranium concentrations in several alluvial aquifer wells upgradient of the Bluewater site varied from 0.003 to 0.04 mg/L. For the purposes of this study, the background uranium concentration in the Rio San Jose alluvial aquifer is assumed to be approximately 0.01 mg/L (Section 5.3).

9.4.3 San Andres Aquifer

Ambient water quality of the San Andres aquifer was evaluated using groundwater sample results reported by the USGS (Baldwin and Anderholm 1992) for 12 wells near Thoreau, New Mexico, approximately 19 mi northwest (upgradient) of the Bluewater site. Much of the groundwater in the aquifer is oxidized, and specific conductivity values indicate moderately high salinity. All of the samples were undersaturated with gypsum, indicating that the groundwater has a tendency to dissolve gypsum; gypsum dissolution is a potential cause of the moderately high salinity.

The San Andres aquifer upgradient of the Bluewater site is not impacted by contaminants, as no uranium mining or milling activity occurred in upgradient areas. Molybdenum was not detectable at the wells near Thoreau, and background uranium concentrations in the bedrock aquifer were no greater than 0.011 mg/L. For the purposes of this study, the background uranium concentration in the San Andres aquifer is assumed to be approximately 0.01 mg/L, which is the same value estimated to represent background concentration in the alluvial aquifer (Section 5.3).

9.5 Performance of the Bluewater Disposal Cell

Increasing uranium concentrations in DOE alluvial aquifer well T(M) between 2001 and 2010 raised a concern about whether a pulse of contaminated fluids was leaking from the main tailings disposal cell. The performance of the disposal cell was investigated to determine if such a pulse had occurred. Anaconda and ARCO documents were examined to characterize the history of tailings and leachate disposal at the main tailings disposal cell, and to better understand its design and construction. A combination of site observations, radon measurements, and review of studies on other disposal cells and landfills was used to evaluate the performance of the cell and its cover.

The historical characterization revealed that several billion gallons of tailings fluid (leachate) seeped through the bottom of the main tailings impoundment and into the underlying alluvial and San Andres aquifers prior to construction of the disposal cell cover in 1995 (Section 6.2). ARCO expended considerable effort to dewater the tailings prior to placement of the cell cover, but expected seepage to continue until the tailings completely drained. ARCO assumed that the cell cover would prevent infiltration of precipitation.

Other cell and landfill cover studies indicate that precipitation typically infiltrates and percolates through the covers of rock-covered cells. This finding suggested that infiltration of precipitation on the main tailings disposal cell provided a continuous source of recharge. The rates of seepage into the cell as a result of precipitation were estimated using estimated hydraulic properties for cell cover materials, the tailings, and strata underlying the tailings (Section 6.4). It appears that an undetermined portion of the tailings remains unsaturated and is considered available for storage of water derived from precipitation on the cell cover. The calculations used to derive estimated cell inflows and available storage indicated that seepage through the base of the cell is likely occurring, but at very low rates at this time. Additionally, the calculations indicated that downward seepage from tailings into underlying strata would increase as the depth of saturation in the tailings increases. It was assumed that current seepage losses from the cell are contributing to the underlying mineralized zone that apparently acts as a continuing source of groundwater contamination. However, the volume of tailings fluid and contaminant mass leaving the cell under current conditions are estimated to be very small, especially in comparison to the volume and mass that seeped through the bottom of the tailings impoundment during milling operations (Section 6.5).

Depressions have formed on the disposal cell cover because the slimes portion of the tailings impoundment continued to consolidate after the cover was completed. Runoff produced by precipitation on the cover forms ponds in these depressions, potentially introducing a second source of fluids that can infiltrate the disposal cell. However, the persistence of the ponds and the results of radon flux measurements indicate that there has been no reduction in the performance of the radon barrier and that the ponded water is reduced primarily through evaporation rather than infiltration. Therefore, the depressions and associated ponds are not considered a source of fluids that could seep from the cell (Section 6.5).

Vegetation is gradually establishing on the disposal cell cover, including deep-rooted shrubs that indicate the presence of moisture in the tailings. If the vegetation establishment is allowed to continue, evapotranspiration will reduce the volume of precipitation seeping into the tailings, thus delaying or completely preventing saturation of the full thickness of the tailings (Section 6.3.3). Accordingly, seepage losses from the tailings to underlying strata would be expected to remain minimal. However, it is possible that deep-rooted vegetation may degrade the performance of the cell's radon barrier.

Because no unusual seepage from the disposal cell appears to be occurring, other causes were considered to explain the increasing uranium concentrations in alluvial aquifer well T(M). Uranium concentrations in the other alluvial wells downgradient of the cell have exhibited uranium concentrations that have remained steady or are slightly declining. SRNL (2014) evaluated the aqueous-phase chemistry of the groundwater sampled at well T(M) and concluded that the increasing uranium concentrations at the well were not related to a pulse release of fluid from the disposal cell. The bottom of the well screen is at the contact between the alluvium and the underlying Chinle Formation, and the bottom of the well's borehole is 9 ft below the contact. SRNL reasoned that, as the water level dropped in the well to elevations near the base of the screen, most the sampled water was probably drawn from the Chinle Formation instead of the alluvial aquifer. Though the reason for high uranium concentrations in Chinle Formation water has not been verified, SRNL presented a plausible geochemical explanation. It is concluded in this conceptual model that the increasing uranium concentrations in well T(M) were not caused by a pulse release of contaminated water from the main tailings disposal cell (Section 8.2.2).

9.6 Mechanisms for Mobilization of Uranium and Other Constituents

Uranium is mobilized in the alluvial and San Andres aquifers primarily via dissolution by groundwater of solid-phase uranium in the mineralized zone beneath the Bluewater site's main tailings disposal cell (Section 8.1.1). Some of the solids are assumed to be uranium-bearing minerals that precipitated out of solution from downward-seeping tailings fluids as the acidity of the fluids was progressively neutralized with seepage depth. Additional solid-phase uranium may be adsorbed to rock and alluvium beneath the cell. The solid-phase uranium is likely present in basalt, ancestral alluvium, and the San Andres Limestone.

Though uranium and other constituents might be released to groundwater by downward-seeping water from the cell into underlying unsaturated sediments, contaminant contributions from such a source are likely minor in comparison to available contaminant mass in the mineralized zone. The mineralized zone appears to be a continuous contaminant source that produces uranium concentrations in onsite monitoring wells that are either relatively constant or decreasing very slowly (Section 8.1.1). It is assumed that this source will remain indefinitely.

9.7 Migration and Distribution of Uranium Contamination

9.7.1 Alluvial Aquifer

Examination of potential groundwater velocities in the ancestral river alluvium and uranium concentrations in onsite alluvial aquifer wells leads to the conclusion that uranium transport in this aquifer was rapid. This conclusion comports with previous studies of the Bluewater site that focused on characterization of groundwater flow and transport, which found that the plumes for conservative (nonreactive) constituents reached the east boundary of the site just a few years after initial construction of the main tailings impoundment. The rapid movement of uranium is attributed to fast advective transport in coarse sands and gravels.

Much of the rapid contaminant transport appears to occur in the main paleochannel that has been identified in ancestral river alluvium beneath the south half of the Bluewater site. Under the assumption that the first lava flows of the Bluewater Basalt preserved the channel of the Rio San Jose as it existed just prior to the lava incursion, the mostly coarse deposits in the channel were prevented from being redistributed by temporally varying locations of the river and its floodplain across the river valley. In contrast, former overbank areas likely contain a mixture of coarse- and fine-grained materials due to reworking of those sediments by river flows in years preceding the lava incursion. Comparatively low velocities are expected in these floodplain deposits due to the presence of abundant silts and clays.

The resulting groundwater movement and contaminant transport in the ancestral river alluvium can be characterized as a dual-domain system, wherein the paleochannel is a preferential flow path conveying groundwater and contamination downstream at a high velocity, and remaining groundwater contamination in overbank areas migrates much more slowly, converging on the channel from both of its sides (Section 8.4.7). Accordingly, contamination in the overbank deposits flushes much more slowly from the groundwater system, at rates that are similar to those caused by molecular diffusion. This process of "back diffusion" leads to an overall retardation of contaminant transport from the ancestral river portion of the alluvial aquifer, such that complete flushing of the aquifer is potentially delayed by tens to hundreds of years.

As previously discussed, uranium migrating southeastward in the ancestral alluvium paleochannel from the Bluewater site merges with westward-moving uranium contamination from the Homestake site about 1 mile southeast of the Bluewater site. The historical and current effects of the merged plumes on groundwater in areas farther downgradient of the Bluewater site, such as in the vicinity of Milan, are unclear. Very little information exists for historical concentrations of uranium and other constituents in that portion of the alluvial aquifer underlying Milan. However, it can be deduced that the volume of flow and the amount of contaminant mass contributed by the alluvial paleochannel passing under the Homestake site has historically been much greater than the flow and contaminant mass contributed by the Rio San Jose paleochannel (Section 8.4.4). Uranium concentrations exceeding the drinking water standard of 0.03 mg/L have been reported for alluvial aquifer wells as far south as Toltec, but the processes leading to these concentrations have not been evaluated.

A large portion of the alluvial aquifer from about a mile north of Toltec to Grants appears to directly overlie the San Andres Limestone. If historical groundwater pumping from the San Andres aquifer in this region reversed the vertical hydraulic gradient from upward to downward, as discussed in preceding sections on the conceptual model, it is possible that contaminated alluvial groundwater reached the bedrock aquifer in earlier years. Though uranium concentrations in samples collected from the Milan municipal wells have been safely below the drinking water standard of 0.03 mg/L, groundwater from the northernmost well has exhibited concentrations of 0.013 mg/L, which is slightly above the adopted background concentration of 0.01 mg/L.

9.7.2 San Andres Aquifer

The apparent presence of Bluewater site-derived uranium at wells in the vicinity of the Homestake site as of 1980 is indicative of rapid east-southeastward advective transport in the San Andres aquifer from the main tailings disposal cell. This is attributed to high average linear velocities associated with conduit flow in karst features in the San Andres Limestone and possibly to quick groundwater migration through fractures within sandstone strata. The much slower velocities and mostly vertical transport in unfractured rock are reflective of dual-domain transport (Section 8.4.7), with advection dominating in the mobile domain (secondary permeability features) and mostly diffusive transport dominating the immobile domain (unfractured rock). Accordingly, the uranium that diffuses into the aquifer matrix during early stages of plume development can be expected to back-diffuse into secondary permeability features as uranium concentrations in the aquifer gradually decline. The exchange between domains effectively retards uranium transport in the aquifer, resulting in a plume that persists much longer than would occur if transport took place solely within fractures, solution channels, and cavities.

Figure 63 shows the current estimated uranium plume and associated flow paths in the San Andres aquifer. This graphic is an idealized representation of the plume based on recent hydraulic head and uranium concentration data. In contrast to the smooth concentration contours used to depict the plume, the spatial distribution of uranium in the aquifer is more likely to be irregular, particularly because most of the groundwater flow takes place in secondary permeability features as opposed to a porous medium. Correspondingly, the plume has the potential to bifurcate in some locations, such that uranium migration takes place in relatively thin, isolated plumes as opposed to a single, spatially continuous plume. Nonetheless, as illustrated Figure 63, the leading and lateral edges of the uranium plume, as defined by a background concentration of 0.01 mg/L, are estimated to be more than a mile north of Milan and Grants.

9.8 Physicochemical Phenomena that Influence Contaminant Transport

Because groundwater in the alluvial and San Andres aquifers is oxidized, precipitation of solid-phase uranium via microbially mediated reduction is not expected to impact the migration and spatial distribution of the contaminant. This signifies that attenuation of the uranium plumes in the respective aquifers via chemical reduction cannot be relied upon either as a groundwater remedy or a mechanism for containing the uranium within Bluewater site boundaries.

It does appear that adsorption of uranium to mineral surfaces in both the alluvial and San Andres aquifers is occurring. However, the degree to which adsorption retards the migration of uranium in the two groundwater systems is not known. It is likely that hydrous ferric oxide is present as a potential sorbent in both aquifers, but virtually no information is available regarding its abundance. The same holds true for additional sorbent minerals, such as those composing clays in the alluvial aquifer. Though retardation of uranium due to sorption processes has not been adequately characterized, it is likely that dual-domain transport phenomena in both aquifers are causing some contaminant retardation.

Though the effects of chemical reactions, adsorption, and dual-domain processes cannot be fully characterized at this time, sampling results from wells located downgradient of the main tailings disposal cell at the Bluewater site indicate that uranium effectively attenuates with transport distance in both the alluvial and San Andres aquifers (Sections 8.2 and 8.3). This attenuation is currently attributed to transverse dispersion (transverse mixing) processes in both horizontal and vertical directions (Section 8.4.2). Highly irregular flow patterns in karst features in the San Andres aquifer are expected to enhance transverse dispersion in that aquifer, whereas substantial heterogeneity of fluvial deposits is assumed to be the primary cause of horizontal transverse dispersion in ancestral Rio San Jose alluvium. It is also possible that transient flow phenomena in either aquifer lead to a form of "apparent" transverse dispersion. In addition to transverse mixing processes, apparent attenuation in the alluvial aquifer can also be attributed to dilution of the uranium plume by recharge from uncontaminated sources, such as inflow of fresh groundwater, seepage from surface-water features, and infiltration of applied irrigation water.

Inspection of uranium concentration histories at onsite alluvial aquifer wells reveals that, in addition to concentrations decreasing with transport distance, the concentrations have tended to either remain constant or slowly decrease with time (Section 8.2.2). Similar observations are made regarding the San Andres aquifer by examining temporal plots of uranium contamination at onsite wells east of the main tailing disposal cell and wells in the vicinity of the Homestake site (Section 8.2.3). These findings support the observation that the mineralized zone beneath the main tailings disposal cell represents a continuous, constant-concentration source of tailings-related contaminants.

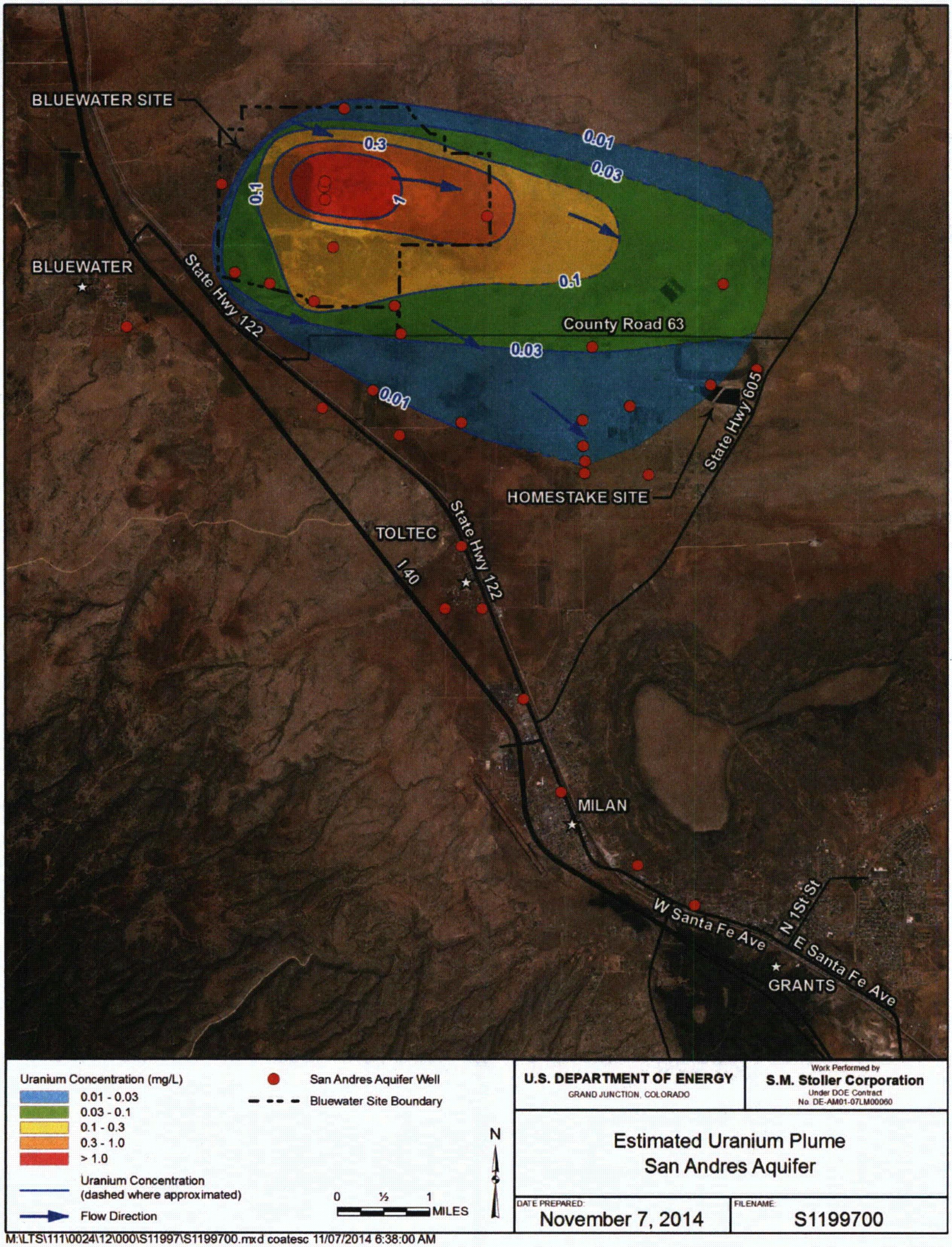

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Figure 63. Current Estimated Uranium Plume in the San Andres Aquifer

9.9 Fate of Uranium

The combination of a continuous, constant-concentration contaminant source, rapid advective transport, and attenuated concentrations with transport distance has produced a uranium plume in the San Andres aquifer that has remained relatively stable since at least 1980. Comparison of mapped uranium plumes in the aquifer for four different time periods (1980–1981, 1996–1997, 2008, 2008–2014) reveals a plume whose length and width have remained virtually constant for more than 30 years (Section 8.3.1). Extending from the main tailings disposal cell to areas in the vicinity of the Homestake site, the leading edge of this plume, as defined by a uranium concentration of 0.01 mg/L, is estimated to lie in an area east-southeast of the Homestake site and north of Grants.

The development of a uranium plume in the San Andres aquifer fed by a continuous source and characterized by mostly steady concentrations due solely to the effects of transverse mixing may seem counterintuitive. However, such a plume becomes possible when it migrates sufficiently far downgradient that uranium concentrations continually produced along the leading and lateral plume edges are at background levels (0.01 mg/L).

Uranium concentration histories at San Andres aquifer wells east of the main tailings disposal cell suggest that the mineralized zone beneath the cell will remain a continuous, constant-concentration source for the foreseeable future. Consequently, the uranium plume in the aquifer is not expected to change significantly for at least several decades, if not hundreds of years. This signifies that uranium contamination originating at the Bluewater site will not affect municipal water supplies in Milan and Grants. Nor is it likely that a pulse of contamination from the disposal cell will affect the uranium plume in future years.

9.10 Potential Risk to Downgradient Groundwater Users

Samples from alluvial aquifer wells monitored by HMC downgradient of the location where the ancestral Rio San Jose aquifer merges with the San Mateo Creek aquifer have uranium concentrations that exceed the New Mexico drinking water standard of 0.03 mg/L. However, these concentrations have not been observed in municipal supply wells operated by Milan. Annual reports on the progress of aquifer remediation activities conducted as part of the GRP at the Homestake site suggest that uranium concentrations in the alluvial aquifer are gradually decreasing.

The municipal water supply wells for Bluewater, Milan, and Grants pump groundwater from the San Andres aquifer. Reported concentrations of uranium at those wells since as early as 1996 have all been less than the drinking water standard. These data, combined with the finding that the uranium plume in the San Andres aquifer lies east of Bluewater and more than a mile north of Milan and Grants, indicate that groundwater contamination originating in the aquifer at the Bluewater site does not currently pose a risk for community water systems in the Grants-Bluewater Valley. The relatively stable uranium plume in the aquifer suggests that future impacts on the municipal wells in the region are unlikely.

None of the private drinking water supply wells in the vicinity of the Bluewater site sampled by DOE have uranium concentrations exceeding the drinking water standard. However, not all of the well owners contacted by DOE have allowed their wells to be sampled. Though the unsampled wells appear to lie outside existing groundwater flow paths emanating from the Bluewater site, it cannot be stated with certainty that no local domestic wells have been adversely impacted by Bluewater-derived contamination.

9.11 Summary of Groundwater Conceptual Model

The major risk-related components of the groundwater conceptual model are summarized below. Figure 64 shows a stylized cross section of the San Andres aquifer uranium contamination and risk.

- Seepage from the carbonate and main tailings impoundments during mill operations contaminated the underlying alluvial and San Andres aquifers. Uranium concentrations exceed the New Mexico drinking water standard in both aquifers at the downgradient site boundary.
- The seeping fluids were neutralized in the materials beneath the cells and apparently formed a mineralized zone in the basalt, alluvium, limestone, and sandstone materials and in fault zones. This mineralized zone is considered to be a continuing source of contamination in both aquifers as groundwater flows through it.
- Minimal seepage from both disposal cells is likely occurring because of residual tailings fluids within the cells and infiltration of precipitation through the cell covers. However, no surge of tailings fluids into the aquifers has occurred since milling operations ceased in the 1980s.
- Contaminated alluvial aquifer groundwater from the Bluewater site merges with contaminated alluvial groundwater from the Homestake site about 1 mi downgradient of the Bluewater site. The combined flow continues in the southeast direction toward Milan.
- San Andres aquifer groundwater beneath the Bluewater site flows in the east-southeast direction toward the Homestake site and areas north of Grants. Contaminated groundwater apparently reached the Homestake site no later than 1980.
- Groundwater remediation by HMC has reduced contaminant concentrations in the alluvial aquifer. However, many alluvial wells downgradient of the Homestake site have uranium concentrations exceeding the drinking water standard.
- The leading edge of the San Andres aquifer uranium plume is apparently downgradient of the Homestake site and has changed little since the early 1980s. The uranium plume in the aquifer is influenced by multiple processes, including a continuous source of uranium contamination in the mineralized zone beneath the main tailings disposal cell and transverse dispersion processes that help maintain essentially stable concentrations upgradient of the leading edge of the plume.
- No alluvial or San Andres aquifer domestic wells solely affected by Bluewater site-derived contamination have uranium concentrations exceeding the drinking water standard.
- Collectively, the direction of groundwater flow in the San Andres aquifer, the apparent stability of the uranium plume in the aquifer, and the locations of municipal supply wells indicate that Bluewater site-derived uranium contamination does not pose a current or future risk for community water systems in the Grants-Bluewater Valley.

1. Precipitation infiltrates disposal cell cover
2. Continued minor seepage from cell
3. Mineralized zone under cell from tailings seepage
4. Groundwater flows through mineralized zone and mobilizes uranium
5. Uranium concentration decreases with distance from mineralized zone; plume is stable
6. No domestic wells currently in plume area
7. Leading edge of plume a short distance downgradient of Homestake Site
8. Milan and Grants municipal wells unaffected by uranium plume (not in aquifer flow path)

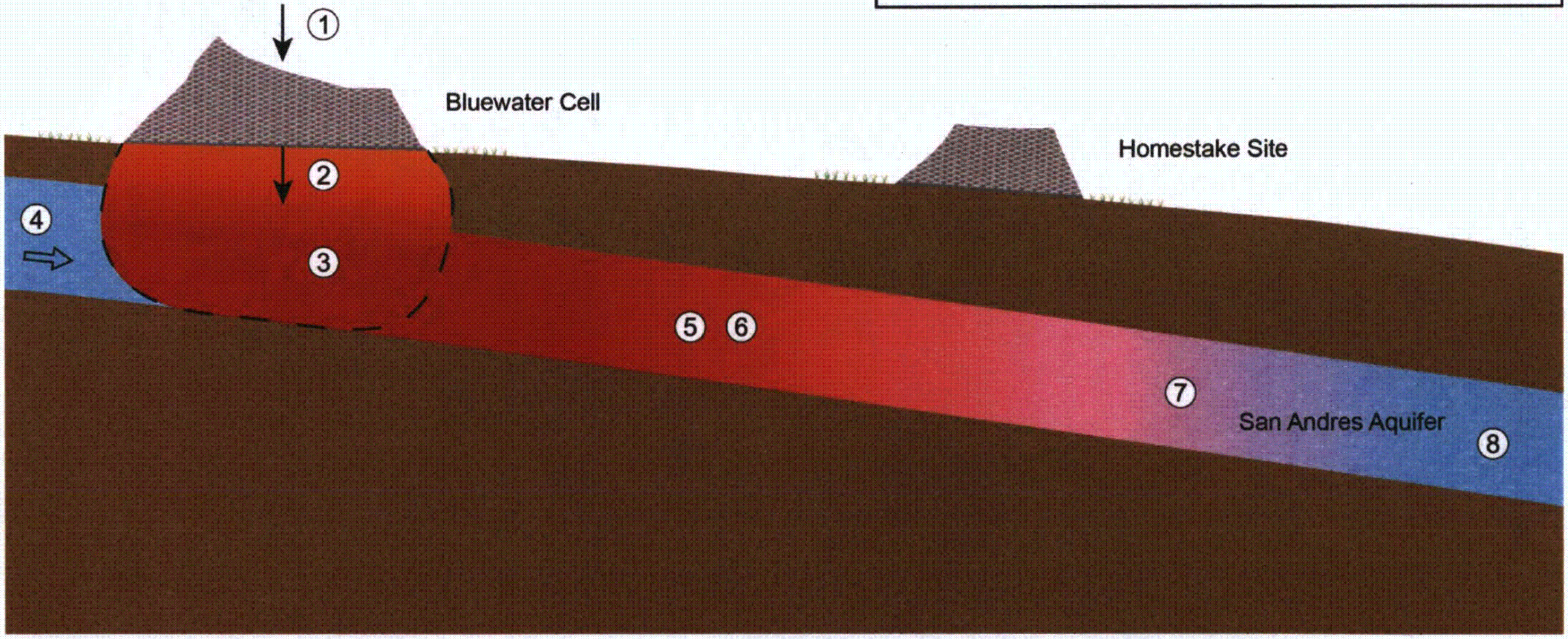

Figure 64. Stylized Cross Section of San Andres Aquifer Uranium Contamination and Risk

10.0 Uncertainties

Uncertainty in the groundwater conceptual model is unavoidable given the sparseness of available data to characterize the hydraulic and chemical properties of the aquifers and the limited amount of information available to corroborate or refute alternative models. Table 17 addresses the major uncertainties associated with the observations and conclusions presented in Chapter 9.

Table 17. Uncertainties and Their Effects on Study Conclusions

Conclusion	Uncertainty	Effect on Conclusion	Significance of Uncertainty
Sources of Contamination			
An estimated 5.7 billion gallons of fluids seeped through the bottom of the Bluewater main tailings impoundment prior to encapsulation. Unknown proportions seeped into the underlying aquifers.	ARCO estimated this quantity based on a water balance analysis of the milling processes. Actual seepage rates and quantities were not measured.	Precipitation and evaporation were not factored into ARCO's estimates. Therefore, actual seepage could have been greater or less than 5.7 billion gallons.	Attempts to more precisely estimate the amount of seepage that occurred would not impact the conclusion that a very large volume of contaminated tailings fluid seeped into the geologic materials and aquifers below the tailings impoundment.
Based on historical groundwater monitoring results near the Bluewater main tailings impoundment, the fluids that seeped through the bottom of the tailings impoundment are assumed to have formed a mineralized zone in the geologic strata and fault zones under the disposal cell.	Sampling of materials under the disposal cells has never been conducted to confirm the presence or extent of the mineralized zone.	Continued elevated uranium concentrations in wells near the disposal cell indicate the presence of a continuing source of contamination. Because ARCO did not observe changes in groundwater chemistry during dewatering activities of the tailings impoundment, there is a high probability that a mineralized zone is present and that it is the primary source of continuing contamination.	Extensive borehole material analysis and groundwater quality measurements under the disposal cells and along the fault zones would be required to define the postulated mineral zone. The characterization most likely would confirm the presence of the mineralized zone and that it should be considered to be a continuing source of contamination to the aquifers. However, it could not completely delineate the extent of the mineralized zone.
The assumed mineralized zone will remain indefinitely as a continuing source of contamination for both aquifers.	It is possible that the source concentrations will decline as inflowing fresh groundwater gradually reduces the contaminant mass. It is also possible that they may increase if groundwater levels rise into the contaminated aquifer matrix materials that are not currently saturated.	If source concentrations decline, then the leading edge of the uranium plume would retreat. Conversely, if source concentrations increase, the leading edge could advance.	With the exception of alluvial well T(M), uranium concentrations in the site wells have remained steady or have declined slightly since the 1980s, after the greatest quantity of seepage from the tailings impoundment occurred. If source contamination declines, it is expected to occur slowly, and a corresponding reduction in the uranium plume would be difficult to discern based on the existing regional well network. It is unlikely that uranium concentrations will increase to levels higher than occurred during milling, so a surge of increased uranium is not expected.

Table 17 (continued). Uncertainties and Their Effects on Study Conclusions

Conclusion	Uncertainty	Effect on Conclusion	Significance of Uncertainty
Seepage continues to occur through the bottom of the Bluewater main tailings disposal cell and is expected to continue indefinitely. However, because of steady or declining uranium concentrations and decreasing water levels in wells near the disposal cell, seepage is assumed to be occurring at a minimal rate. Therefore, the disposal cell is not considered to be a significant source of contamination in the aquifers compared to the source generated and released during milling.	The seepage rate is dependent on the amount of precipitation that percolates through the cell cover, hydraulic characteristics of the tailings materials, and the degree of saturation within the tailings, all of which are unmeasured. Therefore, the current seepage rate is unknown.	Even though the rate is unknown, seepage is most likely occurring. However, the contaminant mass available from present day seepage to impact groundwater contaminant concentrations is extremely small compared to the mass associated with the fluids that seeped during milling.	Measuring precipitation percolation rates, hydraulic properties of the cell materials, and the degree of saturation within the cell would require extensive sampling and testing of the cover and tailings materials. The source term released during milling is much greater than the source term being released from the disposal cell. Therefore, knowing the actual seepage rate would not have a significant bearing on conclusions relating to the behavior of contaminant migration in the aquifers.
Elevated uranium concentrations in DOE's alluvial well T(M) do not represent a new pulse of contamination from the Bluewater disposal cell. The elevated concentrations appear to represent Chinle Formation groundwater.	Although comparisons of historical and current groundwater quality from well T(M) samples suggest that the current uranium concentrations do not represent tailings fluids, the source of the uranium has not been confirmed.	If the elevated uranium in well T(M) is representative of tailings fluid, it may be impossible to determine if it is the result of a new pulse of cell seepage or increased mobilization of uranium from the mineralized zone.	Chemical tests could be conducted in well T(M) that may determine the source of uranium. However, uranium concentrations have not increased in alluvial wells downgradient of well T(M), so fully understanding the cause of increasing uranium in well T(M) would not affect conclusions regarding contaminant transport from the Bluewater site.
Groundwater Flow and Transport			
Groundwater flow and contaminant migration in the San Andres aquifer are influenced by faults to some degree. The faults at the Bluewater site were important conduits for distributing contaminated fluids that leaked from the tailings impoundment into the alluvial and San Andres aquifers at the site.	Because very few faults in the region have a surface expression beyond the foot of the Zuni Mountains, the subsurface locations of the faults are only approximated based on well log information. Therefore, it is difficult to evaluate the degree of their influence on the aquifers. In particular, the effects of pumping for the GRP and by municipal, industrial, and agricultural wells near Milan on flow and contaminant transport through the San Mateo faults are unknown.	Regionally, groundwater flow and contaminant transport in the San Andres aquifer is in the east-southeast direction from the Bluewater site and does not appear to be significantly influenced by faults. Locally, however, faults could direct flow and contaminant transport perpendicular to or vertically from the regional direction. Existing or future San Andres production wells in fault zones could also redirect flow and contaminant transport toward the wells contrary to the regional flow direction, as occurred when the Anaconda production wells were pumping.	Depending on the amount of vertical displacement and the formations involved, faults can behave as conduits or barriers to groundwater flow, or may not influence regional flow. The Milan municipal wells are completed in or near San Mateo fault zones, so additional geochemical analyses or other tests may be useful to help understand potential contaminant migration through the fault zones toward those wells. However, the Milan municipal wells do not show upward trends in uranium concentrations, so additional study may not be warranted at this time.

Table 17 (continued). Uncertainties and Their Effects on Study Conclusions

Conclusion	Uncertainty	Effect on Conclusion	Significance of Uncertainty
Groundwater flow and associated contaminants in the Rio San Jose alluvium merge with flow and contaminants in the San Mateo Creek alluvium about 1 mi downgradient of the Bluewater site. The merged alluvial aquifer flows southeast toward Milan.	The hydrologic and chemical processes that occur in the merge area have not been evaluated, so the contribution of Bluewater-derived contaminant mass to the alluvial system downgradient of the merge area is unknown.	Attenuation of uranium in the Rio San Jose alluvial groundwater appears to be significant by the time its flow merges with contaminated San Mateo Creek alluvial groundwater. Also, uranium concentrations in the Rio San Jose alluvial groundwater are substantially lower than those in the San Mateo Creek alluvial groundwater. Consequently, flow and contaminant transport are likely dominated by the San Mateo Creek alluvial system. Regardless of the contribution of flow and contaminants from the Bluewater site, uranium concentrations exceeding the drinking water standard are present in the alluvial groundwater at least as far as Toltec.	Aquifer tests in both alluvial systems upgradient of the merge zone and a geochemical evaluation of sample data from existing wells in both systems and within the merge zone would be required to evaluate hydrologic and chemical processes occurring in the merge zone. Although this information would be helpful in understanding the alluvial aquifer systems, it would not change the observation that contaminated groundwater has migrated at least as far as Toltec, which is upgradient of Milan. Also, remediation efforts at the GRP are expected to reduce Homestake-derived contaminant concentrations that enter the merge zone.
Uranium concentrations attenuate with distance from the contaminant source. The groundwater in the alluvial and San Andres aquifers is in an oxidized state, so chemical reduction of uranium is not a process influencing attenuation of uranium in the aquifers. Therefore, dispersion is most likely the main process by which attenuation occurs.	Insufficient data are available to characterize the presence of adsorbent minerals (e.g., hydrous ferric oxide) or local reducing environments in the aquifers.	The presence of attenuation processes other than dispersion within the aquifer systems would be advantageous in limiting the extent of the uranium plumes.	Additional analysis of historical and new groundwater sample results, and possibly analysis of core data from new wells along the flow paths, may lead to a better understanding of attenuation processes within the aquifers. However, determining the actual processes of attenuation is not considered critical because the contaminant plumes are stable.
Bluewater-derived contamination in the San Andres aquifer apparently reached the Homestake site by 1980. It is assumed that elevated uranium concentrations in San Andres aquifer wells at and near the Homestake site (including HMC-928 well north of the Homestake site) are attributed to Bluewater-derived contamination, and that the uranium plume between the Bluewater and Homestake sites is continuous.	There is a 2.5-mi gap between the sites without San Andres aquifer wells; therefore, it is difficult to validate the assumptions. ARCO's deep injection well on the Bluewater site contributed to San Andres aquifer contamination, but the magnitude and persistence of the contamination associated with that well are unknown. Also, the contribution of contaminants into the San Andres aquifer resulting from GRP operations and pumping from the Milan municipal wells is unknown.	If elevated uranium concentrations in San Andres aquifer wells near the Homestake site are due partially or wholly to GRP operations, then contaminants have not migrated as far from the Bluewater site as expected.	DOE has an NRC-approved LTSP and is committed to expend resources whenever a clear nexus to radiological safety is established. DOE resources were used to prepare this site status report and groundwater conceptual model, and DOE will maintain a continuing dialog with NRC and NMED for ways to improve our common understanding of the groundwater flow and contaminant transport in the Grants-Bluewater Valley.

Table 17 (continued). *Uncertainties and Their Effects on Study Conclusions*

Conclusion	Uncertainty	Effect on Conclusion	Significance of Uncertainty
San Andres aquifer flow and uranium transport processes between the Homestake site and Grants are assumed to be similar to those between the Bluewater and Homestake sites.	No San Andres wells are present between the Homestake site and Grants, so flow and uranium transport processes in that region are unknown.	The leading edge of the uranium plume (the 0.01 mg/L contour) could be farther advanced toward areas north of Grants than currently estimated.	Uranium concentrations in the Grants municipal supply wells do not show effects of mill-related contamination. DOE is committed to expend resources whenever a clear nexus to radiological safety is established, and will maintain a continuing dialog with NRC and NMED for ways to improve our common understanding of the groundwater flow and contaminant transport in the Grants-Bluewater Valley.
Potential Risk to Groundwater Users			
Assuming current San Andres aquifer use remains the same, the Milan and Grants municipal water supply wells will continue to have uranium concentrations below the drinking water standard.	Pumping from high-production municipal, industrial, and irrigation wells could influence regional flow patterns in the San Andres aquifer, but the degree to which this pumping could influence flow is unknown.	Pumping from the Anaconda production wells altered San Andres flow patterns; the natural flow direction has recovered since pumping ceased. It is possible that increased pumping south of the estimated uranium plume could draw San Andres aquifer groundwater and its contaminants to the south where it could impact municipal supply wells.	To date, pumping south of the plume appears to have little effect on regional groundwater flows. However, if declines in regional San Andres water levels continue, and/or pumping increases, there is a possibility of an adverse effect on the municipal groundwater supply.
Although uranium concentrations in Milan's municipal wells are expected to remain below the drinking water standard, water in their northwestern-most well appears to be impacted by mill-related contaminants. Uranium concentrations are greater than the adopted background concentration of 0.01 mg/L, and the U-234/U-238 activity ratio may suggest the presence of processed uranium.	Insufficient analyses have been conducted to verify the presence of processed uranium in the Milan well water.	If processed uranium is present in the Milan well water, it would be difficult to determine the source of the uranium. Pumping by the Milan wells and other San Andres aquifer wells in the area appears to have reversed the hydraulic gradient between the alluvial aquifer and the San Andres aquifer in the vicinity of the Milan wells. If this has occurred, then the processed uranium could be derived from the contaminated San Mateo Creek alluvial aquifer as alluvial water is drawn down into the San Andres aquifer by pumping.	Although slightly above background, the uranium concentrations in Milan's water supply wells are well below the drinking water standard and have not shown upward trends; therefore, the water is safe to drink. If the GRP is successful and if uranium concentrations in the Milan municipal wells remain steady or decline, then additional analyses of the hydrology and hydraulics of the aquifers in the vicinity of Milan's wells may be unnecessary.

Table 17 (continued). Uncertainties and Their Effects on Study Conclusions

Conclusion	Uncertainty	Effect on Conclusion	Significance of Uncertainty
Permitted private drinking water wells in the vicinity of the Bluewater site that were sampled by DOE do not have uranium concentrations exceeding the drinking water standard.	DOE did not sample all wells in the immediate downgradient vicinity. Some well owners declined DOE's sampling request, and some owners did not respond to DOE's request. Furthermore, there is a possibility that there are unpermitted wells in the area used for drinking water.	Unsampled private wells could have Bluewater-derived contaminants exceeding the drinking water standard.	No permitted alluvial wells used for drinking water purposes are located between the Bluewater site and the zone where the ancestral Rio San Jose aquifer merges with the San Mateo Creek aquifer. If uranium contamination is present in wells downgradient of that zone, it may be difficult to discern the source of contamination. No known San Andres wells permitted for drinking water use are present within the estimated uranium plume. If a drinking water well were to be installed within the plume area, it would likely encounter Bluewater-derived uranium.

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Appendix A

Appendix A

Water-Balance Assessment for the Bluewater Main Tailings Impoundment and Disposal Cell, Bluewater, New Mexico

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LMS/BLU/S10666

Water-Balance Assessment for the Bluewater Main Tailings Impoundment and Disposal Cell Bluewater, New Mexico

June 2014

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LMS/BLU/S10666

Water-Balance Assessment for the Bluewater Main Tailings Impoundment and Disposal Cell
Bluewater, New Mexico

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Abbreviations

ACL	alternate concentration limit
Anaconda	Anaconda Copper Company
ARCO	Atlantic Richfield Company
cm	centimeters
cm/s	centimeters per second
ΔS	change in storage
DOE	U.S. Department of Energy
ft	feet
gpm	gallons per minute
LiDAR	light detection and ranging
LTSP	Long-Term Surveillance Plan
mg/L	milligrams per liter
NMED	New Mexico Environment Department
NRC	U.S. Nuclear Regulatory Commission
PCBs	polychlorinated biphenyls
pCi/m^2s	picocuries per square meter per second
POC	point of compliance
POE	point of exposure
UMTRCA	Uranium Mill Tailings Radiation Control Act

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Executive Summary

The Bluewater, New Mexico, uranium mill operated from 1953 to 1982. Site reclamation was completed in 1995 and included encapsulating the main tailings impoundment in place in an engineered, rock-covered disposal cell. The U.S. Department of Energy (DOE) acquired the site in 1997. The Bluewater cell is the largest disposal cell currently managed by DOE.

Site groundwater monitoring conducted by DOE has indicated that uranium concentrations exceed groundwater standards in the two uppermost aquifers at the site. DOE is evaluating the source of the contamination and the potential risk to downgradient groundwater users. This assessment is intended to provide an understanding of how much mill processing fluid and contaminant mass drained from the main tailings impoundment prior to completion of the disposal cell, and if the tailings in the disposal cell are a continuing source of contamination.

Atlantic Richfield Company (ARCO), the former licensee, estimated that approximately 5.7 billion gallons of tailings fluid seeped through the bottom of the main tailings impoundment prior to construction of the disposal cell cover in 1995; about half of that total was projected to have occurred prior to 1960. These fluids entered the two uppermost aquifers at the site—an alluvial aquifer beneath basalt lava flows, and the deeper San Andres aquifer. ARCO expended a considerable effort to dewater the tailings before completing the disposal cell, so the volume of tailings liquor available for continued seepage was significantly reduced.

Tailings fluids, consisting of a less-contaminated mixture of the remaining tailings liquor and precipitation that has percolated through the cover materials, are seeping from the disposal cell and may continue to do so indefinitely. Projected maximum estimates for annual seepage, based on saturated conditions within the cell and a potential upper limit of 50 percent of precipitation percolating through the cell cover, are substantially less than 1 percent of the total seepage that occurred prior to 1995. However, current seepage appears to be minimal because decreasing water levels and steady-state contaminant concentrations in the aquifers are not indicative of the maximum estimated rate. It is unlikely, therefore, that the tailings are saturated at this time, and hydraulic properties of the cover materials may not have changed enough to allow the maximum projected precipitation infiltration. An increase in vegetation on the cell cover is expected to keep infiltration rates low due to evapotranspiration, which would keep seepage rates low.

Nearly 1 million pounds of uranium may have been present in the tailings fluids that seeped from the main tailings impoundment. Monitoring of the aquifers by ARCO indicated neutralization of the acidic tailings fluids and substantial reduction of contaminant concentrations in groundwater in the area adjacent to the tailings impoundment. These precipitated and adsorbed contaminants presumably formed a mineralized zone in the aquifer materials under the disposal cell. This assumed mineralized zone, rather than seepage from the cell, is suspected to be the main continuing source of groundwater contamination.

Depressions have formed on the disposal cell cover because the clay-rich ("slimes") portion of the impounded tailings continued to consolidate after the cover was completed. Precipitation runoff water forms ponds in these depressions, potentially introducing a second source of fluids infiltrating through the disposal cell. However, observations of the persistence of the ponds and the results of radon flux measurements indicate that there has been no reduction in the performance of the radon barrier and that the pond volumes are reduced primarily through evaporation rather than infiltration. Therefore, the ponds are not causing additional seepage from the cell.

This assessment found no evidence to suggest that the elevated uranium concentrations in onsite wells indicate a new pulse of seepage or contamination from within the disposal cell or from ponding on the cell. Uncertainties associated with the water balance and contaminant mass estimates do not negate the primary conclusion that the volume of fluid and mass of contaminants that might seep from within the cell since it was constructed are very small compared to the volume and mass that seeped through the bottom of the tailings impoundment prior to cell construction.

1.0 Purpose of Assessment

The U.S. Department of Energy (DOE) Office of Legacy Management manages the Bluewater, New Mexico, Uranium Mill Tailings Radiation Control Act (UMTRCA) Title II Disposal Site. The site is located in Cibola County (formerly Valencia County) in west-central New Mexico. DOE manages the site in accordance with a Long-Term Surveillance Plan (LTSP), approved by the U.S. Nuclear Regulatory Commission (NRC), to ensure protection of human health and the environment. Requirements of the LTSP (DOE 1997) include verifying that the disposal cell functions as designed and that groundwater meets approved quality standards.

Site groundwater monitoring conducted by DOE has indicated that uranium concentrations exceed groundwater standards in the two uppermost aquifers at the site. DOE is evaluating the source of the contamination and the potential risk to downgradient groundwater users. This assessment is part of that evaluation and is intended to provide an understanding of how much mill processing fluid and contaminant mass entered the aquifers prior to completion of the disposal cell, and if the disposal cell is a continuing source of contamination.

A uranium mill operated at the site for 30 years, processing ore from offsite mines in the region. The site was reclaimed, and the two tailings impoundments at the site were encapsulated as the carbonate tailings and main tailings disposal cells. The two uppermost aquifers at the site have uranium concentrations substantially above background concentrations. It has been assumed that the contamination of the aquifers occurred during milling operations and prior to tailings impoundment encapsulation. However, the performances of the disposal cells have come into question because of increasing uranium concentrations in a site monitoring well and the development of depressions that collect precipitation runoff on the main tailings disposal cell cover.

The former licensee recognized that substantial quantities of tailings fluids leaked through the bottom of the main tailings impoundment and into the underlying aquifers. Therefore, this assessment focuses on the main tailings impoundment and subsequent disposal cell. Most of the tailings generated during mill operations were deposited in this impoundment, and most of the impoundment performance and groundwater-related research conducted by the former licensee dealt with this impoundment.

Although significant quantities of tailings fluids were injected into a deep well in the northeast portion of the site, and later collected in large evaporation ponds north of the main tailings pond, these features are not addressed in this assessment. The injection well was located more than a mile northeast of the main tailings impoundment, and the tailings fluids were injected into a formation underlying the site aquifer of concern (the San Andres aquifer). A former monitoring well that was located near the injection well was sampled by the mill operator and indicated contamination of the San Andres aquifer; however, the magnitude of upward leakage into the San Andres aquifer and the extent of contamination were not evaluated at the time. The potential impacts of the deep-well injection process on the San Andres aquifer will be addressed in a groundwater conceptual model being developed by DOE. The evaporation ponds, constructed northeast of the main tailings impoundment, were lined and therefore were not considered to be a significant source of seepage. Furthermore, the native materials under the pond locations are not in direct contact with the aquifers.

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2.0 Site Description

2.1 Site Description and History

The Bluewater site, approximately 9 miles northwest of Grants, New Mexico, is the location of a former uranium-ore processing site that operated from 1953 to 1982. The uranium mill was constructed and operated by the Anaconda Copper Company (Anaconda). In 1977, Anaconda was acquired by the Atlantic Richfield Company (ARCO).

Milling operations created radioactive mill tailings that were stored in two onsite tailings impoundments. ARCO began decommissioning the mill in 1989 and began site reclamation in accordance to a plan (ARCO 1990) approved by NRC. By 1995, all mill tailings, contaminated soils, demolished mill structures, and contaminated vicinity property materials were encapsulated in six onsite disposal areas (ARCO 1996). Mill tailings were capped in place in two locations, referred to as the carbonate tailings disposal cell and the main tailings disposal cell. Responsibility for the site was transferred to DOE in 1997. The site, shown in Figure 1, is administered under the provisions of an NRC general license.

The most significant site feature is the main tailings disposal cell. The completed disposal cell covers an area of 354 acres, and the enclosed original tailings impoundment has a footprint of approximately 260 acres (two smaller extensions were added during site reclamation). An estimated 22.9 million tons of tailings are in the disposal cell, which makes it the largest disposal cell managed by DOE. The next largest cell is approximately 7.1 million tons at the Falls City, Texas, UMTRCA Title I Disposal Site.

2.2 Site Hydrogeology

2.2.1 Geology

Site geology plays a significant role in the groundwater contamination issues. Figure 2 shows a geologic map of the site. A large portion of the site's surface consists of basalt lava. A series of lava flows eventually filled the former channel of the Rio San Jose. The basalt, which covers alluvial sands and gravels of the former Rio San Jose, has an average thickness of approximately 100 feet (ft) at the site. The buried alluvium, ranging up to 25 ft in thickness, is partially saturated and comprises the uppermost aquifer at the site.

The remainder of the site surface is composed of a small outcrop of the Permian San Andres Limestone; shale, siltstone, and sandstone members of the Triassic Chinle Formation; and Quaternary alluvium consisting primarily of windblown sand, silt, and clay. The Permian Glorieta Sandstone formation underlies the San Andres Limestone and, therefore, does not crop out at the site. These two formations have a combined thickness of approximately 250 ft at the site.

Although deeply buried under most of the site, a portion of the San Andres Limestone is exposed on the surface because of the complex structural geology of the site. Two inactive but significant faults crisscross the site at an approximate 90 degree angle. The north-south fault, referred to regionally as the Ambrosia Lake Fault, tracks under the location of the main tailings disposal cell. The unnamed east-west-tracking fault crosses the Ambrosia Lake Fault beneath the southern end of the main tailings disposal cell. These faults are shown in Figure 2, and their projected subsurface configurations are shown in Figure 3 and Figure 4.

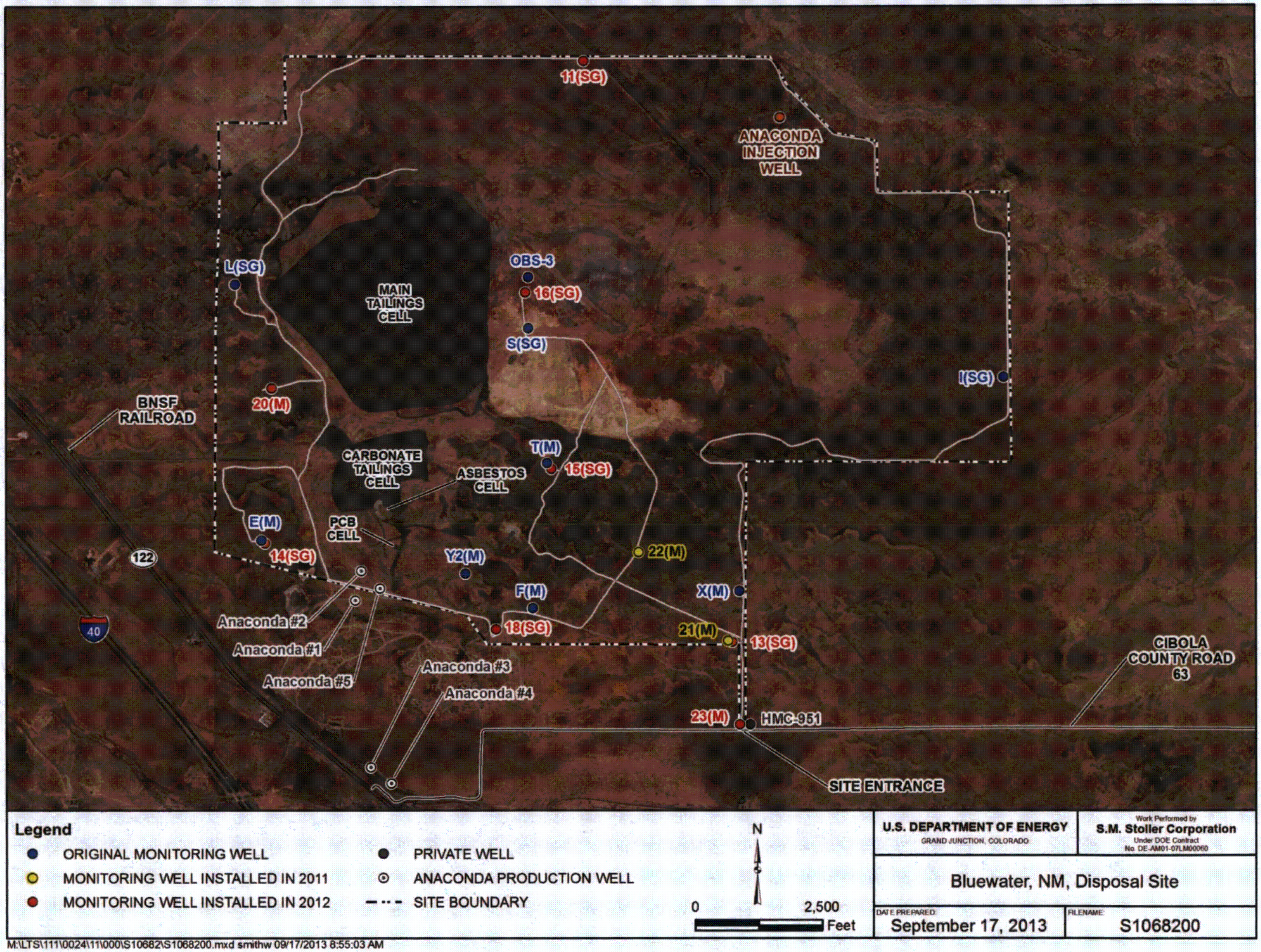

Figure 1. Bluewater, New Mexico, UMTRCA Title II Disposal Site

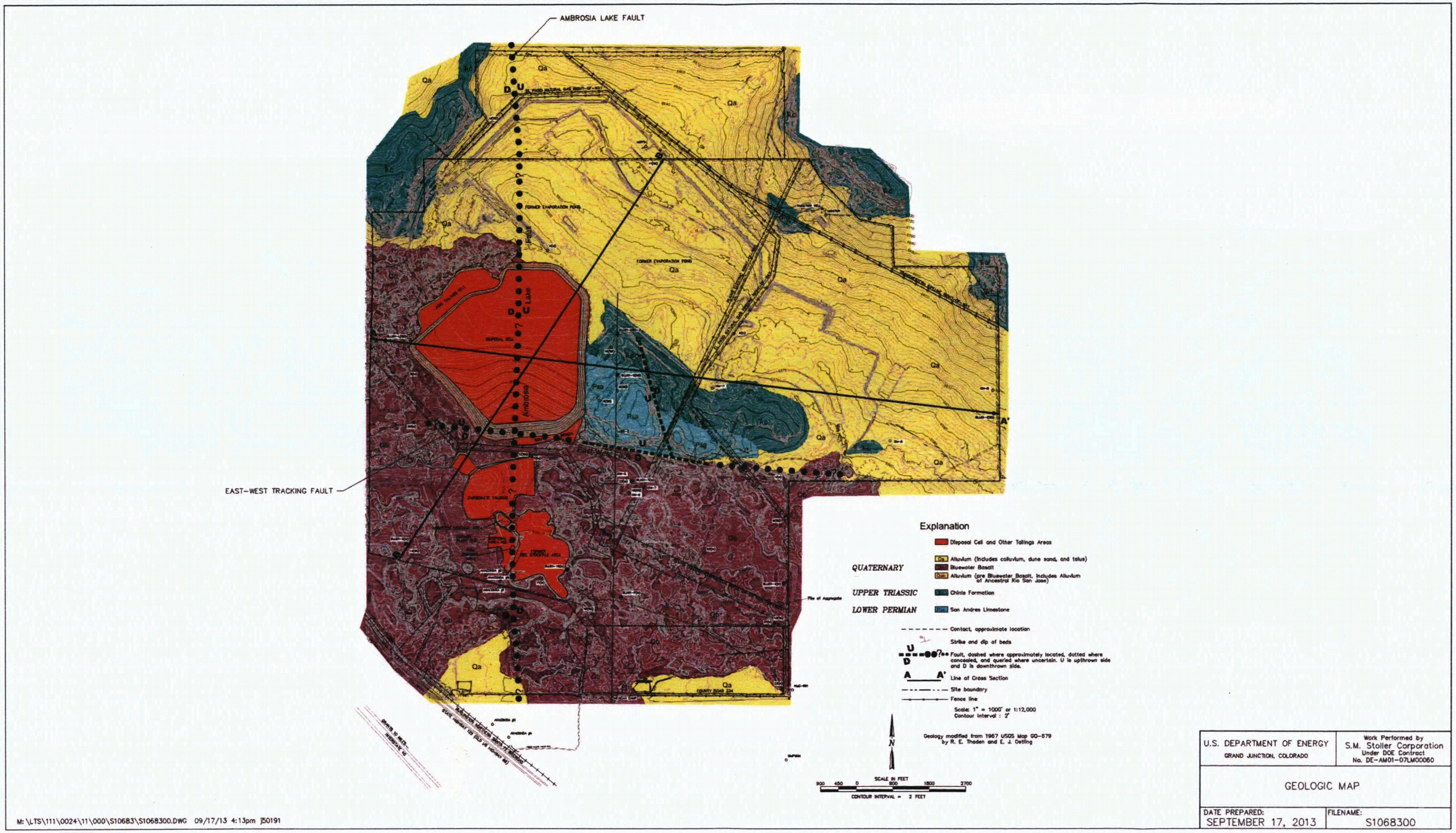

Figure 2. Geologic Map of the Bluewater Disposal Site

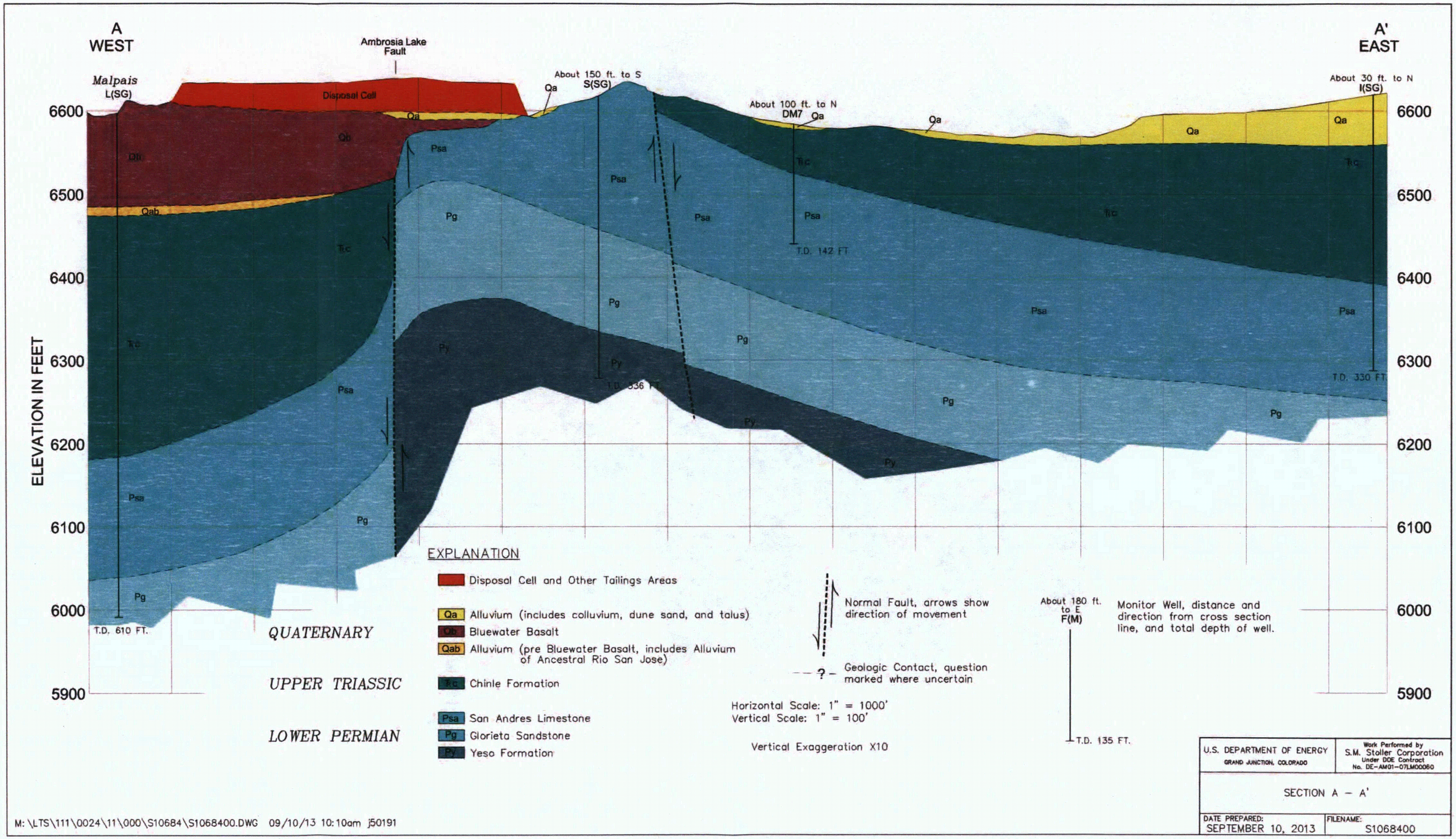

Figure 3. Geologic Cross Section A–A'

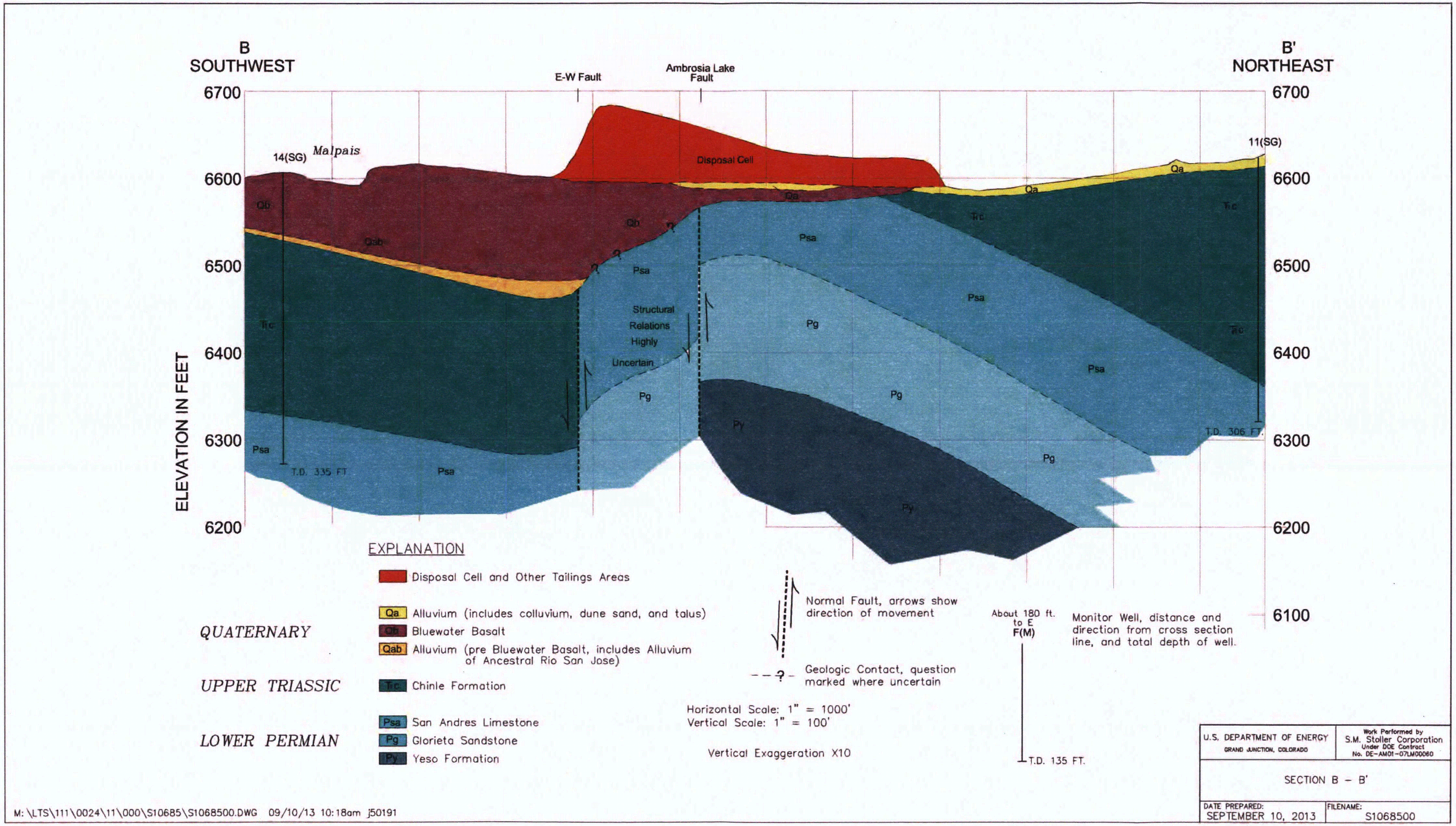

Figure 4. Geologic Cross Section B–B'

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Regionally, the Permian- and Triassic-age formations dip in the northeast direction. This dip, however, is complicated by the faults at the site. The configuration of the faults has resulted in four fault blocks at the site. Surface geology and well completion records indicate that the northeast fault block had the greatest upward displacement, while the northwest block had the greatest downward displacement. Well completion records for wells L(SG) and S(SG) indicate that the relative vertical displacement between these two fault blocks is more than 400 ft. The relative vertical displacement between the northeast fault block and the southeast block is approximately 370 ft. The least displacement, of approximately 50 ft, occurred between the southeast and southwest fault blocks.

The faults do not act as sharp demarcations between the fault blocks. The formations on each side of the faults bent and dragged along the fault as represented in Figure 3 and Figure 4. A characteristic of faulted limestone and sandstone is the resultant zone of shattered rock along the fault (referred to as a fault zone). The shales and siltstones of the Chinle Formation, however, tend to mold like clay along fault zones.

2.2.2 Groundwater Hydrology

The surface Quaternary deposits, located in the northeast portion of the site, are dry and do not behave as an aquifer. Though some thin sandstone members of the lower Chinle Formation are in the subsurface of the site, this formation locally behaves as an aquitard above the underlying San Andres Limestone, and it is not considered to be an aquifer at the site.

The uppermost aquifer at the site is in the basalt-covered sand and gravel alluvium of the ancestral Rio San Jose. This alluvium ranges from dry to fully saturated at the site, depending on location. Because of the difficulty of drilling through the basalt lava and the risk of missing saturated portions of the alluvium, very few wells are completed in this aquifer near the site. The groundwater in this aquifer flows generally southeast across the south portion of the site along the path of the buried river valley.

A deeper aquifer was also contaminated at the site. Regionally and at the Bluewater site, the San Andres Limestone and Glorieta Sandstone formations form a single hydrologic unit referred to as the San Andres aquifer. Although it is confined by the overlying Chinle Formation (and, therefore, exhibits artesian pressure) under most of the site, the exposed portion of San Andres Limestone in the southwest corner of the northeast fault block (adjacent to the main tailings disposal cell) is dry. The San Andres Limestone was partially saturated during milling operations but has become dry at this location because of lowering groundwater elevations. The deeper Glorieta Sandstone portion of the aquifer is saturated at this location. Because of the northeast dip of the formations, the entire San Andres/Glorieta sequence exhibits saturated and artesian pressure conditions between the San Andres Limestone outcrop and the site boundary where the Chinle Formation behaves as a confining layer over the San Andres Limestone. ARCO concluded from San Andres aquifer tests using site wells that the east-west-tracking fault forms a hydraulic boundary that effectively divides the San Andres aquifer into two flow regimes.

The San Andres aquifer is the primary water supply source for municipal, commercial, irrigation, domestic, and livestock use in the region. Although the entire thickness of the aquifer produces groundwater, wells are usually completed in the upper 50 ft because of the high production rates that occur in that portion of the aquifer. The groundwater in this aquifer flows generally east-southeast from the site. The Village of Milan, which uses groundwater from this aquifer, is the closest downgradient municipality (located about 5 miles southeast of the site). The Village of Bluewater, which also uses San Andres aquifer groundwater, is located southwest and upgradient of the site and would not be affected by site-related contamination.

2.3 Historical Groundwater Issues

As part of its reclamation plan, ARCO attempted to clean up the groundwater to background (upgradient) water quality but was unsuccessful because aquifer pumping did not reduce contaminant concentrations. Consequently, ARCO recommended setting alternate concentration limits (ACLs) for the mill-related contaminants that were showing concentrations higher than background concentrations (Applied Hydrology Associates Inc. 1990). NRC, the site regulator, approved ACLs for uranium of 0.44 milligram per liter (mg/L) for the alluvial aquifer point-of-compliance (POC) wells and 2.15 mg/L for the San Andres aquifer POC wells. The New Mexico drinking water standard for uranium at the time of site reclamation was 5 mg/L, so the approved ACLs were substantially below the state standard. The ACL for selenium is 0.05 mg/L for both aquifers. The alluvial aquifer also has an ACL for molybdenum, which is 0.10 mg/L. The ACLs were not expected to be exceeded at the POC wells.

ARCO's groundwater evaluations indicated that, although alluvial and San Andres aquifer groundwater near the main tailings disposal cell would remain contaminated with mill-related constituents, the contaminants would precipitate or otherwise attenuate to concentrations below New Mexico health-based concentration limits before the groundwater left the mill site boundary. In 2004, however, New Mexico adopted the U.S. Environmental Protection Agency drinking water standard for uranium of 0.03 mg/L, which is significantly below the prior standard of 5 mg/L that was in place when the ACLs were established.

Most of the tailings impoundment is north of the east-west-tracking fault. Because of the apparent hydraulic boundary caused by this fault, ARCO assumed that only the north portion of the San Andres aquifer was significantly affected by mill-related contamination. ARCO knew that contaminated groundwater had been drawn to their production wells south of the tailings impoundment, but assumed that incoming fresh water (from upgradient recharge) would mix with contaminated water and dilute contaminant concentrations to acceptable levels. Low uranium concentrations, ranging from 0.004 to 0.014 mg/L during the 1980s, in a downgradient private well supported this assumption (Sabre-Piñon well, currently known as HMC-951). Consequently, continued monitoring of the San Andres aquifer south of the fault was considered unnecessary, and no monitoring wells were left in that portion of the site.

2.4 Groundwater Monitoring by DOE

2.4.1 Required Monitoring

DOE inherited nine ARCO wells that were to be monitored in accordance with the LTSP. Groundwater is monitored at the site as part of DOE's goal to ensure protection of human health and the environment. When DOE began monitoring at the site in 1998, seven of the wells were to be sampled at specified intervals. Two point-of-exposure (POE) wells—one for each aquifer—located at the site boundary were to be monitored only if ACLs were exceeded in POC wells. The LTSP does not specify concentration standards to be met at the POE wells. The original site monitoring wells are shown in Figure 1.

2.4.2 Alluvial Aquifer

The original alluvial aquifer monitoring well network consisted of background well E(M), POC wells F(M) and T(M), and POE well X(M). It also included well Y2(M) to monitor for polychlorinated biphenyls (PCBs) because of an upgradient PCB disposal cell. Concentrations of molybdenum and selenium have remained within the regulatory limits and do not show upward trends. However, uranium concentrations in alluvial POC well T(M) soon began to increase. The uranium concentration in this well exceeded the alluvial aquifer ACL of 0.44 mg/L in 2010. DOE notified NRC and the New Mexico Environment Department (NMED) of this occurrence in accordance with LTSP requirements.

DOE tried to sample POE well X(M) in 2008 in anticipation of the ACL being exceeded in well T(M); however, it was dry. Consequently, DOE installed new alluvial wells 21(M) and 22(M) in 2011 (Figure 1). Well 22(M) is located very close to the former location of ARCO well U(M), and 21(M) is located near the site boundary approximately the same distance from T(M) as is X(M). DOE conducted a downhole video examination of X(M) in 2012, which revealed dry sediment at an elevation several feet above the reported bottom of the well. The sediment was removed, and a sufficient volume of alluvial groundwater was present to allow sample collection.

Although uranium concentrations in the new wells were below the ACL, they were above the current drinking water standard of 0.03 mg/L. An additional alluvial well, 23(M), was installed in 2012 near the site entrance, the farthest onsite downgradient distance from the disposal cells. Upgradient well 20(M) was installed near the west site boundary but in an area that could have been impacted by mill-related contamination. Uranium concentrations in those wells are below the drinking water standard, which suggests that the groundwater contaminant plume in the alluvial aquifer has had a minimal impact at those locations. The water quality of the alluvial aquifer is addressed further in Section 6.2.

2.4.3 San Andres Aquifer

The original San Andres aquifer monitoring well network consisted of background well L(SG), POC wells OBS-3 and S(SG), and POE well I(SG). These wells are located north of the east-west-tracking fault.

Uranium concentrations in the monitoring wells were below the ACL of 2.15 mg/L and were not showing upward trends. ARCO and DOE results for background well L(SG) have been consistently low, averaging about 0.003 mg/L. However, DOE became aware that a private industrial production well (HMC-951), located near the site entrance and completed in the San Andres aquifer, was showing increasing uranium concentrations. DOE did not have any San Andres wells in the south portion of the site that could be monitored to understand why uranium was increasing in that well.

At about the same time, NMED notified DOE of its concern regarding suspiciously low uranium concentrations in the POC and POE wells. Subsequent downhole videos of the wells showed that the slotted well casings in the POC wells were almost completely blocked by iron scale. It is possible that a reducing environment had developed in the wells, and the low concentrations may have been the result of dissolved uranium precipitating onto the iron scale (the cause of low concentrations is still being evaluated). DOE changed from a low-flow sampling method to a higher-flow casing purge method (the method used by ARCO) in these wells; uranium concentrations have increased but still may be affected by the physical and chemical conditions within the corroded well casings.

Uranium concentrations in POE well I(SG) were significantly below ARCO sampling results. The well has solid steel casing down to the top of the San Andres Limestone, and then open borehole into the Glorieta Sandstone. A downhole video indicated that low-flow samples had been inadvertently collected from within the bottom few feet of the casing instead of within the open borehole as intended. Samples are now collected from within the open borehole portion of the well, and uranium concentrations have increased.

Because of a concern that HMC-951 might have been drawing contaminated water from the Bluewater site, and because of the poor condition of the POC wells, DOE installed six new San Andres aquifer wells on the Bluewater site in 2012. Wells 13(SG), 14(SG), 15(SG), and 18(SG) were installed in the south portion of the site. Well 16(SG) was installed between OBS-3 and S(SG). Well 11(SG) was installed near the north property boundary to help DOE understand the groundwater flow direction in the northeast portion of the site.

Sampling of these wells began in November 2012. None of the new wells have contaminant concentrations exceeding ACLs, but downgradient wells 13(SG) and 18(SG) and POE well I(SG) have uranium concentrations exceeding the drinking water standard. Consequently, contaminated San Andres groundwater has migrated beyond the site boundary in the east-southeast direction. The water quality of the San Andres aquifer is addressed further in Section 6.3.

2.5 Source of Contamination

Sand- and clay-rich tailings were pumped in a slurry to the main tailings impoundment during milling operations. ARCO realized that substantial quantities of tailings fluids leaked through the bottom of the impoundment and eventually into the underlying aquifers, especially prior to the use of the deep-injection well and the evaporation ponds.

The alluvial aquifer was contaminated from tailings fluids that seeped through the bottom of the tailings ponds and then through the porous basalt and into the underlying alluvial sand and gravel. The contaminated groundwater would have then flowed in the alluvial aquifer in the southeast direction along the former channel of the Rio San Jose.

Contamination of the San Andres aquifer occurred through more complex conditions. Tailings fluids seeped directly into the San Andres Limestone outcrop that was covered by the southeast portion of the main tailings impoundment. The San Andres Limestone also subcrops under the basalt beneath a portion of the disposal cell (Figure 3 and Figure 4), so tailings fluid in that area would have drained through the basalt and into the limestone. Contaminated alluvial groundwater most likely seeped into the San Andres Limestone where the alluvium and limestone were in contact. Also, seeping tailings fluids and some contaminated alluvial groundwater would have migrated along the fault zones and into the San Andres aquifer.

A further complicating factor was the southward distribution of contaminated San Andres aquifer groundwater. Anaconda had groundwater production wells south of the site that were used to supply the process water for the mill (Figure 1). The cone of depression resulting from pumping these wells would have drawn contaminated water toward the well field. The Ambrosia Lake Fault zone would have acted as a conduit for rapid southern movement of the contaminated groundwater. Water samples collected by ARCO verified that this occurred; uranium and nitrate concentrations well above background concentrations were observed in the Anaconda #5 production well located along the Ambrosia Lake Fault zone about 1 mile south of the main tailings impoundment. As noted in Section 2.3, however, ARCO did not consider San Andres aquifer contamination south of the east-west-tracking fault to be a significant concern.

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