

## 3.0 Enhanced Characterization

### 3.1 Purpose and Scope

Results of the verification monitoring indicated that natural flushing was generally progressing as expected until June 2010, when significant increases in contaminant concentrations were measured in several wells. The June 2010 sampling event was conducted immediately after record flooding of the Little Wind River. During the flood, overbank flow was observed within a large area downgradient of the former mill site. Significant increases in contaminant concentrations occurred in monitoring wells where the flooding occurred. The spikes in contaminant concentrations were attributed to flood waters mobilizing residual contamination in the unsaturated zone (DOE 2011a).

The observations made in 2010 revealed that the existing site conceptual model and groundwater computer modeling did not account for the spikes in contaminant concentrations observed in the surficial aquifer groundwater. Consequently, the site conceptual model needed to be updated and a new groundwater flow and transport model developed to better simulate natural flushing processes. The enhanced characterization work in 2012 was designed to provide additional data to assist in accomplishing these goals.

The purpose of the enhanced characterization was to obtain additional data to further characterize the surficial aquifer (DOE 2012c). Specific objectives of the investigation were to:

- Provide enhanced definition of contaminant plumes, including the location of the centroid of each plume and the extent of groundwater contamination for each contaminant of concern
- Provide a detailed distribution of contaminants for input into the updated groundwater model.
- Provide data that will guide placement of new monitoring wells outside of the contaminant plumes to monitor lateral plume behavior.
- Provide a detailed and updated baseline of groundwater contamination for tracking plume configuration, movement, and size over time. This will be used to assess the progress of natural flushing if this study is repeated in the future.
- Provide information on soil characteristics, including the leachability of uranium.
- Estimate the mass of uranium remaining in the unsaturated zone of the surficial aquifer, which can be used to develop appropriate contaminant source terms in the transport modeling.

### 3.2 Fieldwork Summary

Fieldwork was conducted August 20–29, 2012. It was performed in accordance with the *Work Plan for the Enhanced Characterization of the Surficial Aquifer, Riverton, Wyoming, Processing Site* (Work Plan) (DOE 2012c). Fieldwork consisted of installing 103 boreholes along 9 transects (Figure 26) with a Geoprobe, which is equipment that is used to direct-push steel rods into the shallow soils and surficial aquifer material (Figure 27). Water samples were collected at each location, and soil samples were collected at 34 locations. To optimize the mapping of contaminant plumes (which may have a slightly different configuration for each contaminant),

transects were oriented northeast and southwest approximately perpendicular to the known southeast direction of groundwater flow and to the axis of the known contaminant plumes. Distance between transects was reduced and sampling density increased in the portion of the aquifer near the expected centroid of the contaminant plumes downgradient of the former mill site. The increase in sampling density was designed to enhance definition of the centroid of the plumes and to provide more soil data from the unsaturated zone above the contaminant plumes in areas where the 2010 flood had the largest effect on groundwater contaminant concentrations. Details of the fieldwork are found in the *Enhanced Characterization of the Surficial Aquifer, Riverton, Wyoming, Processing Site, Data Summary Report* (Data Summary Report) (DOE 2013a).

### 3.3 Soil Characterization

#### 3.3.1 Summary of Methods

Soil samples were collected at 34 locations (Figure 26). Geoprobe rods were driven to 5 feet bgs at each location, and two soil samples (0–2.5 feet and 2.5–5 feet) were collected at most locations for a total of 65 samples. Soils typically consisted of a dry, pale-yellowish silt in the top 2 to 3 feet with some near-surface roots, and sand and gravels below the silt. Figure 28 (top) displays a typical soil-core retrieved from the Geoprobe with a dry silt at the top that grades to a moist clayey-silt and then to sand and gravel. Figure 28 (bottom) shows another soil sample in the process of being homogenized prior to placement into a sample bag. Full sample recovery was not obtained in any of the 2.5–5-foot samples, with a maximum recovery of 84 percent in that interval. No recovery was obtained, and therefore samples were not collected, from the 2.5–5-foot interval at three locations (T01-07, T04-12, and T08-02). Soil characteristics were described and recorded for each location and are documented in the Data Summary Report.

Soil samples were analyzed by the Grand Junction Environmental Laboratory using three different tests: batch tests, kinetic tests, and column tests. Methods used in the laboratory tests are detailed in the *Laboratory Analysis of Shallow Sediment Near a Former Uranium Mill: Riverton, Wyoming, Site* (DOE 2013b) (Appendix F). Samples were dried in air and weighed several times during drying to determine the rate of water loss. Dried samples were sieved through a 2 millimeter (mm) sieve. The proportion of less-than-2 mm fraction varied from 20.95 percent to 99.98 percent of the sample. Artificial site water (ASW) containing a composition similar to the Little Wind River was prepared in the laboratory and used as the primary leaching solution for the tests.

Kinetic tests were conducted to determine the agitation time required for uranium to reach a steady-state concentration. These tests were conducted on eight samples (from four locations) also using ASW. Ten aliquots from each sample were tested with end-over-end agitation times in the test ASW of 5 minutes, 15 minutes, 30 minutes, 1 hour, 2 hours, 4 hours, 8 hours, 16 hours, 48 hours, and 96 hours. After agitation of each aliquot, uranium analysis of the test solution was conducted using Environmental Laboratory analytical method AP (U-2), “Uranium Determination by Chemchek” (DOE 2011b). Soil concentrations in micrograms per gram were calculated from the uranium concentration measured in the test solution.



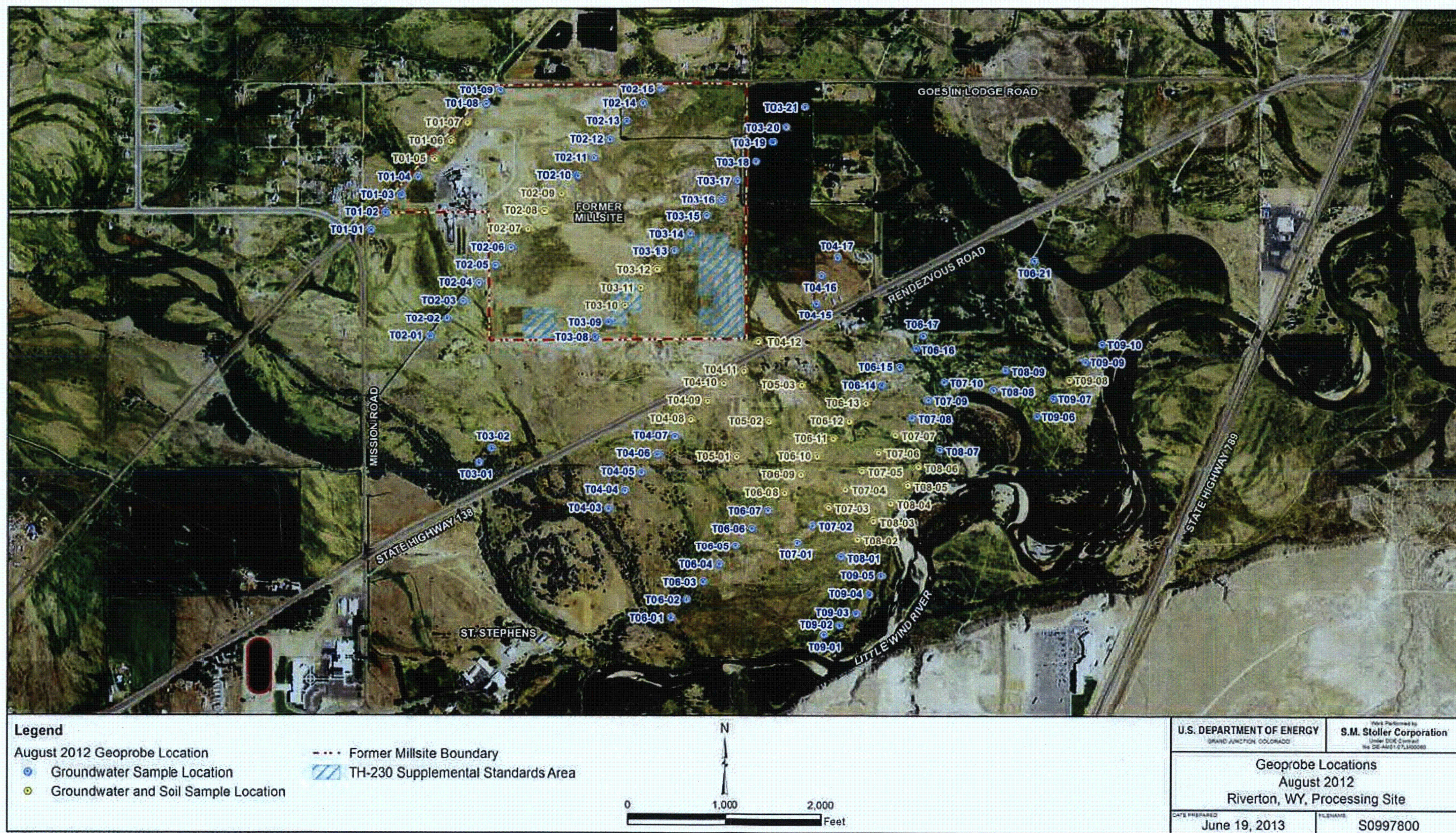


Figure 26. Enhanced Characterization Geoprobe Locations



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Batch tests were conducted on all 65 soil samples by leaching with a high water-to-rock ratio (200 milliliters [mL] of water to 2 grams [g] of soil) using ASW to simulate flood events that would remove uranium. Samples were agitated end-over-end for two separate 24-hour intervals with fresh test solution for each agitation event. Test solutions from the two events were combined and analyzed for uranium using analytical method AP (U-2).



*Figure 27. Installing a Borehole with a Geoprobe in August 2012*

Column tests were conducted on 16 samples (8 locations) to estimate the total uranium source materials remaining in the unsaturated zone. These tests were conducted by pumping ASW through a soil column at a rate of approximately 0.09 milliliters per minute (mL/min) for most columns. Effluent from the column was collected approximately every pore volume and analyzed for uranium using analytical method AP (U-2). Column tests were continued until uranium concentrations in the effluent stabilized, which resulted in completion of tests at various stages ranging from 48 to 93 pore volumes. After the conclusion of the column tests, an extractant solution of sodium bicarbonate ( $\text{NaHCO}_3$ ) and sodium carbonate ( $\text{Na}_2\text{CO}_3$ ) developed by Kohler et al. (2004) was used to extract the easily removable, or labile, uranium from the column sediments.



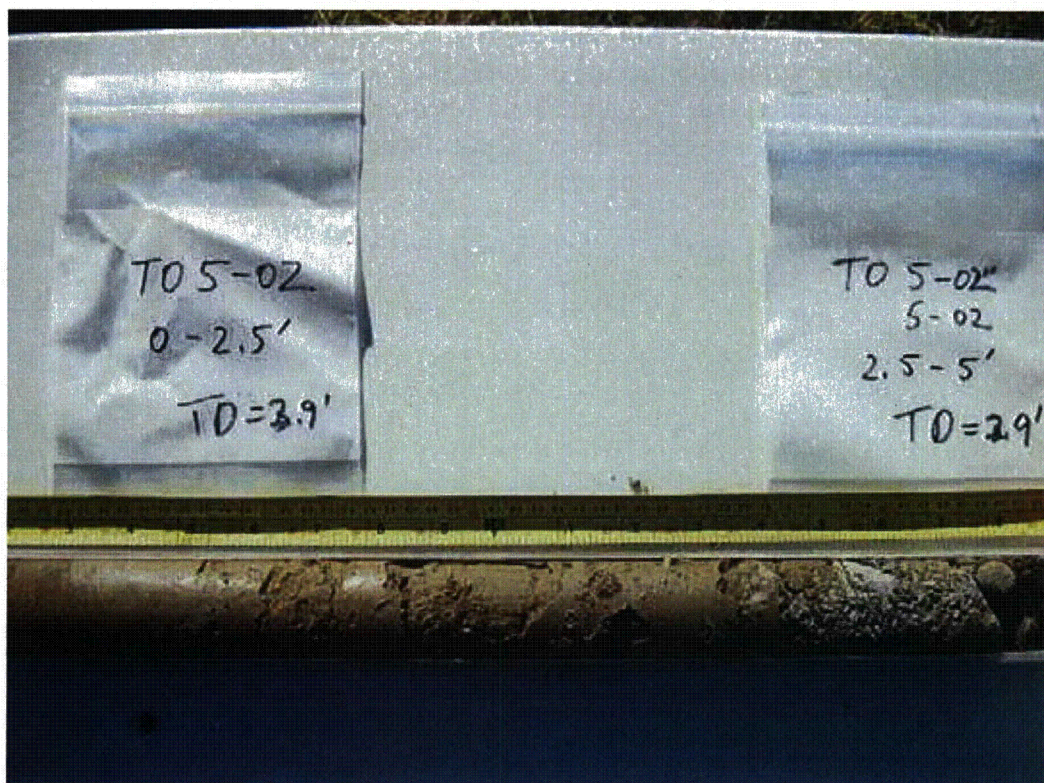


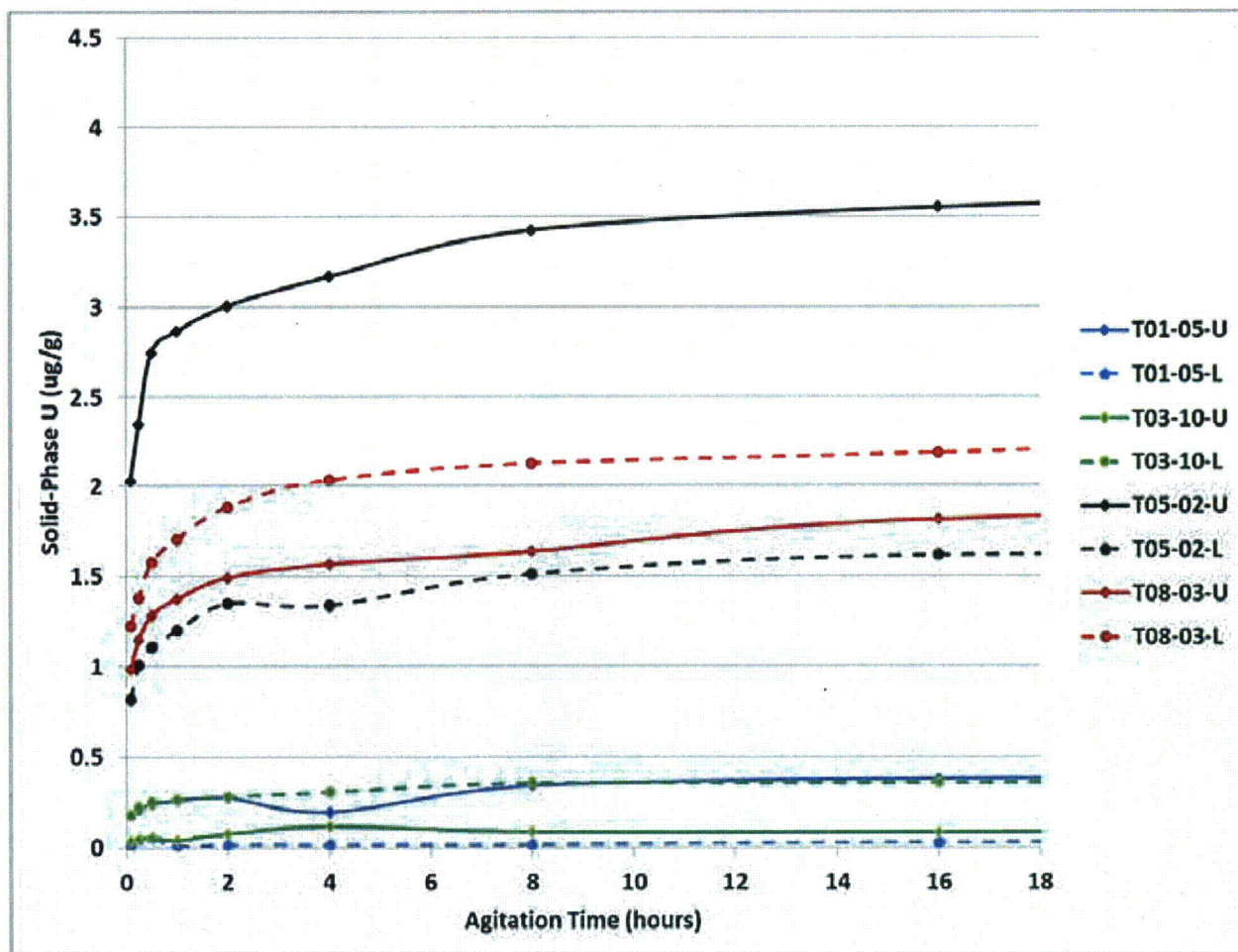
Figure 28. Soil Samples Collected Using the Geoprobe



### 3.3.2 Results and Interpretation

#### 3.3.2.1 Batch Tests

Kinetic test results are shown in Figure 29 and Figure 30. As shown in these figures, the majority of the uranium is removed within the first 8 hours.

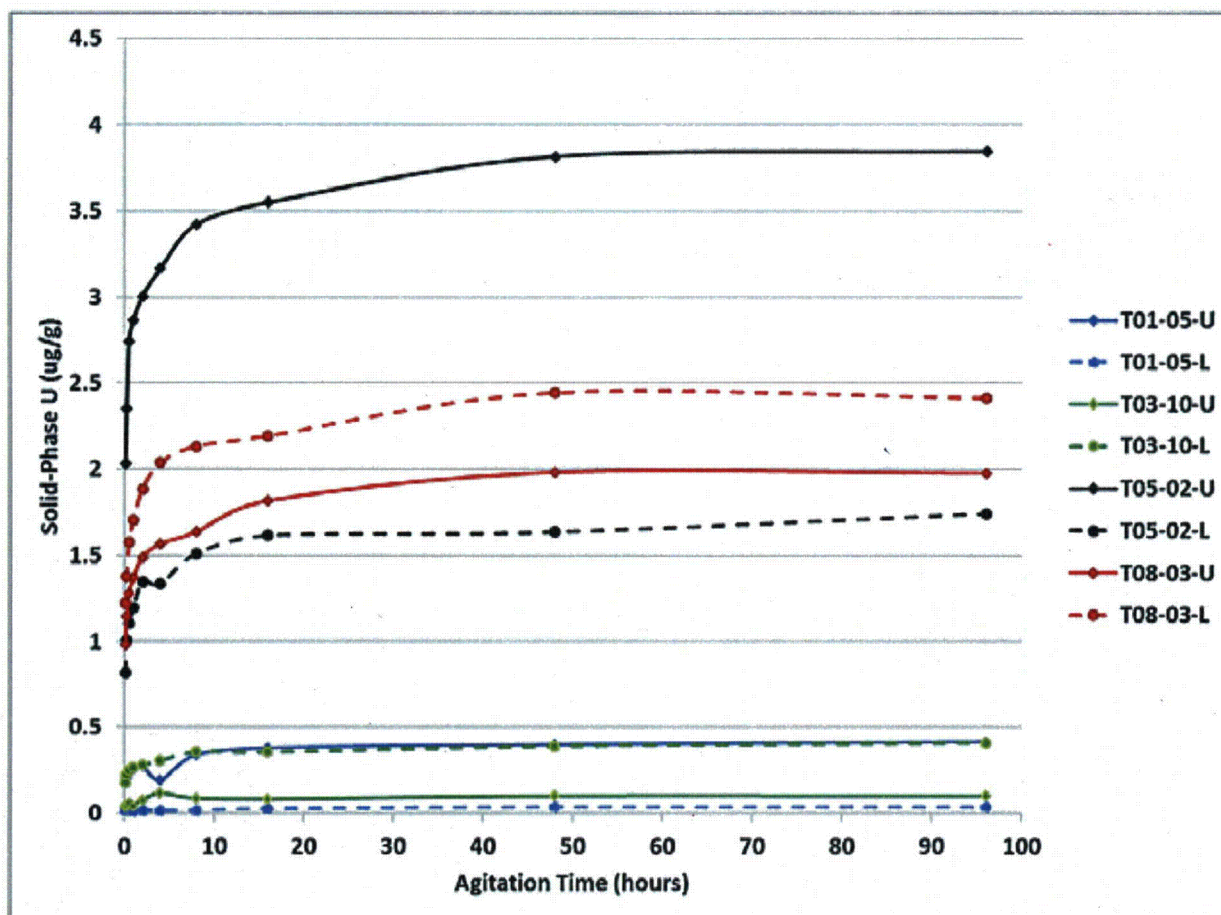


**Abbreviations:**

µg/g = micrograms per gram

U = uranium

Figure 29. Results of Soil Kinetic Tests (0 to 18 Hours)



**Abbreviations:**

µg/g = micrograms per gram

U = uranium

*Figure 30. Results of Soil Kinetic Tests (Full Duration)*

Results of the batch tests indicated a positive correlation between the abundance of fine-grained sediment (<2 mm) and the solid-phase uranium concentrations of the soil samples. The labile fraction is generally considered to be the mass that is weakly sorbed to mineral surfaces and is the fraction that most readily participates in interactions with groundwater. The concentrations of labile uranium measured in the soil samples were comparable to abundances of uranium in sedimentary rocks that make up the crust of the earth.

The concentrations of uranium in the <2 mm sediments that were removed by a 48-hour agitation with ASW were variable, ranging from 0.04 to 4.8 micrograms per gram (µg/g) with a median of 0.96 µg/g (Figure 31). In nearly all paired samples, the upper sample had a higher concentration of removable uranium than the lower sample. Median values for the upper and lower units were 2.10 and 0.34 µg/g, respectively (Figure 31). The distribution of soil concentrations at the Riverton site from batch tests are displayed in Figure 32. The concentrations were generally higher in the offsite (transects 04 through 08) samples than in the onsite (transects 01 through 03) samples.



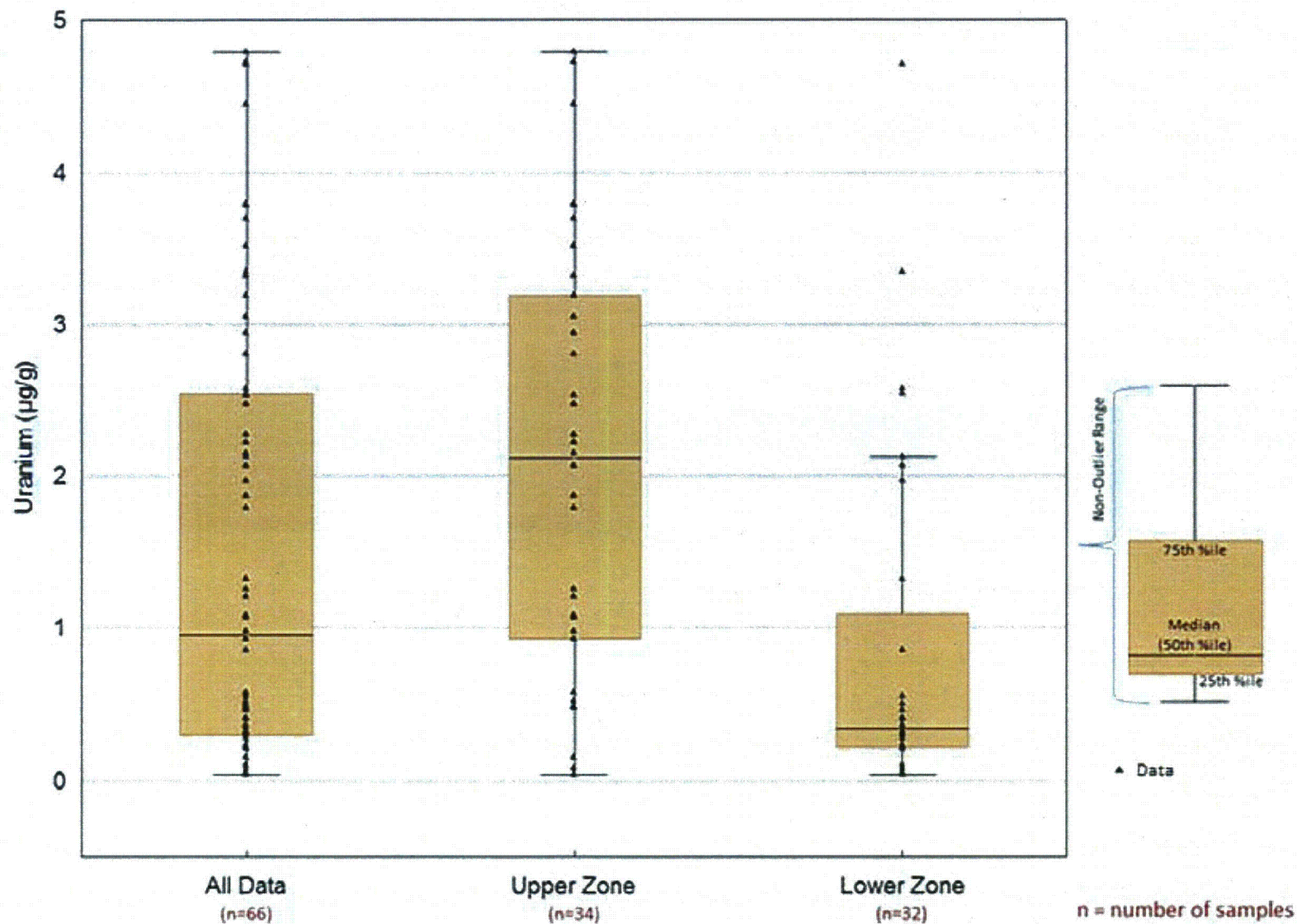


Figure 31. Distribution of Solid-Phase Uranium in Upper Zone (0–2.5 ft) vs. Lower Zone (2.5–5 ft) Samples

### 3.3.2.2 Distribution Coefficients

The transfer of uranium between sediments and groundwater is often modeled by assuming that the ratio of uranium concentration in the sediment to the concentration in the groundwater is constant. The ratio is called the distribution coefficient ( $K_d$ ) and it is often used by groundwater modelers, in part because it is easily incorporated into numerical groundwater codes. It is well known that the actual partitioning of uranium concentrations between groundwater and sediment varies with chemical parameters, in particular the concentration of dissolved carbonate and pH. Thus, groundwater aquifers are likely to display variable  $K_d$  values over space and time. It is also known that the transfer of mass between aquifer solids and groundwater is controlled to some extent by rate-limited processes, such as slow diffusion from immobile pore fluid. These rate-limited processes are not typically considered in flow and transport models. Despite these uncertainties, it is instructive to examine the range of uranium  $K_d$  values that might be observed in the subsurface sediments.

The batch test data collected during the enhanced characterization study were not suitable for determining  $K_d$  values on their own because the concentration of labile uranium in the solid phase was not measured. However, assuming that the labile fractions measured on splits of the same samples used in column tests are representative of the labile fractions in the splits used for batch testing,  $K_d$  values can be calculated as shown in Table 6.  $K_d$  values calculated in this way ranged from 4.30 to 158.75 milliliters per gram (mL/g).

Table 6. Uranium  $K_d$  Values Calculated from Batch Test Data and Column Labile Fractions

Sample Number	24 Hour Batch Test Data		Column Labile	After Batch	$K_d$ (mL/g)
	U ( $\mu\text{g/L}$ )	U ( $\mu\text{g/g}$ )	U ( $\mu\text{g/g}$ )	U ( $\mu\text{g/g}$ )	
T01-05U	4.8	0.48	0.657	0.177	36.88
T02-07L	0.8	0.08	0.136	0.056	70.00
T03-10U	0.8	0.08	0.207	0.127	158.75
T03-10L	4.7	0.47	0.715	0.245	52.13
T04-10U	25.4	2.54	3.761	1.221	48.07
T04-10L	5.1	0.51	0.729	0.219	42.94
T05-02L	13.2	1.32	1.921	0.601	45.53
T06-10U	17.9	1.79	2.033	0.243	13.58
T06-10L	2.2	0.22	0.329	0.109	49.55
T07-04U	12.1	1.21	1.262	0.052	4.30
T07-04L	2.4	0.24	0.312	0.072	30.00
T08-03U	20.7	2.07	2.265	0.195	9.42
T08-03L	25.8	2.58	2.716	0.136	5.27

$\mu\text{g/L}$  = micrograms per liter



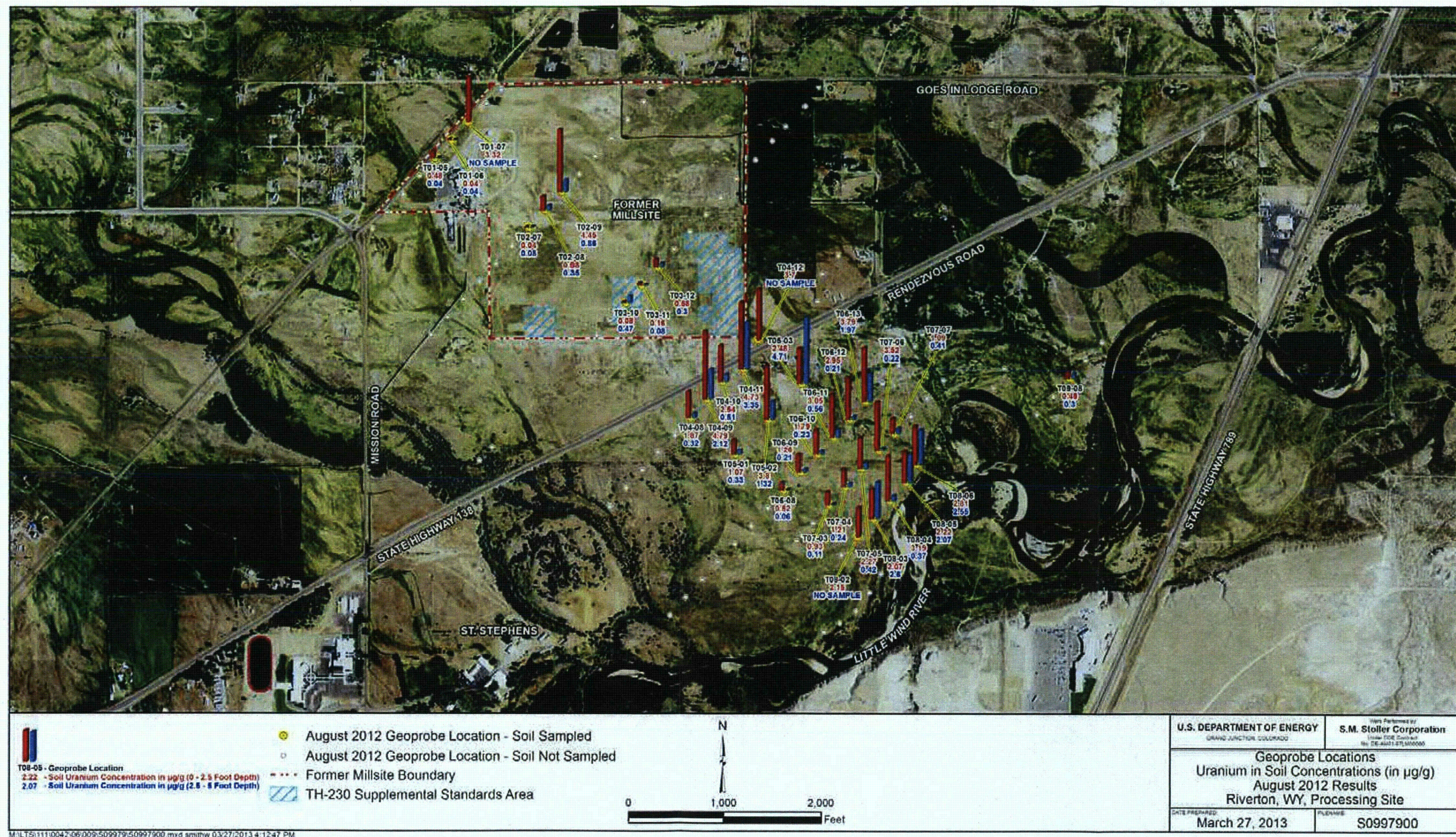


Figure 32. Uranium Distribution from Soil Batch Tests



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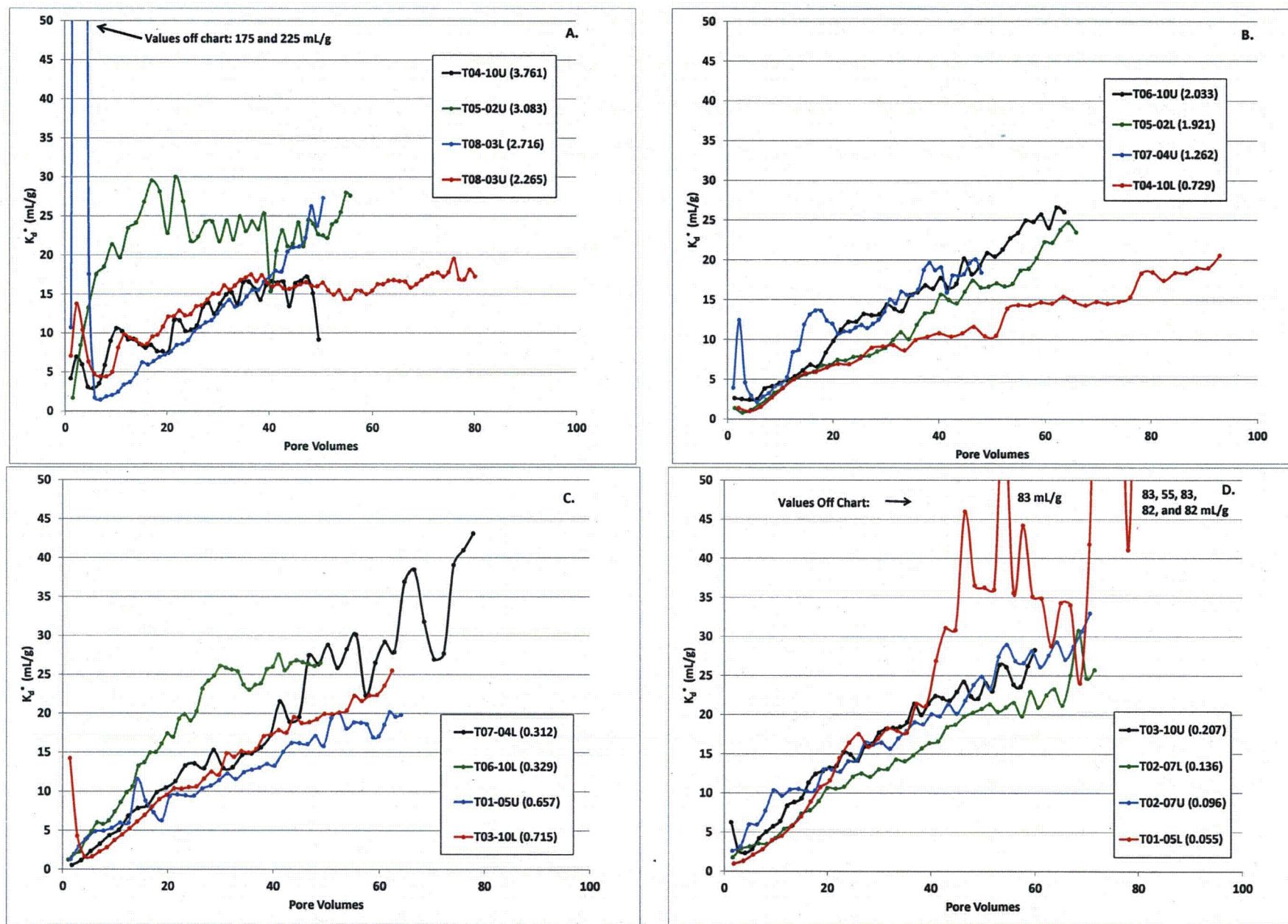
The column data can also be used to estimate  $K_d$  values. The apparent uranium distribution coefficient ( $K_d^*$ ) was determined for each effluent sample collected from the column tests. Because the distribution of uranium in the columns is unknown,  $K_d^*$  is the ratio of the concentration of uranium in the effluent sample to the *average* concentration of uranium in the column sediment. To evaluate *true*  $K_d$ , the concentration of uranium in the column sediment at the top of the column would need to be used with the effluent dissolved concentration. Residence time also influences the estimated  $K_d$  values since the fluid is not in contact with the sediment long enough to come to equilibrium, as is indicated by batch test results measured at variable time intervals.

With the exceptions of fluctuations in the early stages (first 10 pore volumes), the uranium effluent concentrations in the column effluents demonstrated a monotonic decrease throughout the tests.  $K_d^*$  values are plotted on Figure 33, arranged in order of their total labile uranium concentrations in  $\mu\text{g/g}$ .

With only a few exceptions, the  $K_d^*$  values are low early on and gradually trend to higher values as more pore volumes are passed. The fluctuations observed in some columns (e.g., T05-02) during the early stages may be due to fluctuation in the dissolved inorganic and organic carbon concentrations. Some of the fluctuation observed in the later stages (e.g., in T01-05L) are due to imprecision in the uranium concentration analysis at the low dissolved concentrations present in some of the effluent samples.

The early values of  $K_d^*$  are typically less than 5 mL/g, with some as low as about 1 mL/g. Most of the  $K_d^*$  values observed after more than 40 pore volumes exceed 10 mL/g. Numerical models using  $K_d$  values of more than 10 mL/g should result in minimal plume movement (an example is provided in Freeze and Cherry 1979). Because uranium appears to be released and transported by groundwater at the Riverton site, it is likely that the  $K_d^*$  values measured in the column tests do not accurately reflect the nature of the uranium partitioning between sediment and groundwater.

Reasons for the wide distribution of  $K_d^*$  in the column tests are uncertain. Some of the variation is due to mobilization of organic carbon during the early stages of column operation and the possible influence of pH and dissolved inorganic carbon species. Conceptually, the use of the  $K_d$  approach in predictive modeling mandates that the system be maintained at chemical equilibrium. The condition of equilibrium may not be met during the column testing. The residence time of about 1 hour is insufficient for the system to reach an equilibrium state. The condition of equilibrium can be tested using a flow-interruption technique. After a flow-interruption, there should be no change in effluent concentrations if the system is at equilibrium; however, rate-limited reactions are indicated by higher concentrations following column restarts. There are many physical and chemical processes that could cause rate-limited mass transfer. A time lag can occur simply due to slow desorption from mineral surfaces. Slow diffusion from intraparticle pores can also limit uranium transfer. As sediment ages, uranium can migrate to internal portions of mineral crystal structures or to intracrystalline microfractures and pores where it becomes more recalcitrant to re-release. The Riverton sediments have had decades for these types of transformations to occur. Regardless of the exact processes involved, it is reasonable to speculate that the release of contaminants from these sediments is rate controlled.



**Note:** The labile uranium concentrations in  $\mu\text{g/g}$  are given in the parentheses following the sample name—e.g., T03-10U (0.207).

Figure 33. Plot of Apparent Distribution Coefficients ( $K_d$ )

### 3.3.2.3 *Secondary Source in the Unsaturated Zone*

Two scenarios were examined to determine if uranium concentrations measured in soil samples from the 2012 enhanced characterization were high enough to cause the spikes observed in samples collected from monitoring wells in 2010 after flooding of the Little Wind River. These two scenarios were: Scenario 1 – groundwater rising up into the unsaturated zone and mobilizing uranium during flood events, and Scenario 2 – flood waters infiltrating down from the surface and mobilizing uranium. It should be noted that soils collected during the 2012 investigation did not represent the entire unsaturated zone because the samples were only collected from 0 to 5 feet (the unsaturated zone extends deeper most of the year), and recovery ranged from 0 to 85 percent in the 2.5 to 5-foot interval.

Scenario 1 assumes that rising groundwater levels driven by an increasing river stage provide the sole mechanism for leaching of contaminants in the alluvial aquifer's unsaturated zone, and that the resulting leachate was the primary cause of the spikes in concentration observed at near-river wells 0707 and 0789 shortly after a river flood event in June 2010. The viability of this scenario can be assessed by examining flow and advective transport processes associated with this alternative and the results from preliminary modeling of those processes. The observed increase in uranium concentration at well 0707 in response to the 2010 flooding was from about 0.8 mg/L to 2.7 mg/L, and the corresponding increase at well 0789 was from about 1.5 mg/L to 2.64 mg/L (Figure 11).

It is assumed that the rising groundwater elevation was caused by pressure wave transmission inland (toward the Riverton site) from the river, and none of the increase in groundwater level was caused by infiltration of overbank flood water on the floodplain surface. This mechanism had the potential to be most effective at mobilizing contamination if the water table rose as high as the ground surface, so that a maximum amount of sediment was contacted by groundwater. Contaminant mobilization was also likely enhanced by longer contact time between the rising groundwater and the shallow contaminated sediment. The extent of groundwater rise and the contact time depended on aquifer hydraulic conductivity, the specific yield of the aquifer, and the magnitude and duration of the increased river stage during a flood.

Though leaching of uranium from the shallow floodplain sediments apparently increased concentrations in the uppermost part of the saturated zone, additional processes would have been required to transport the newly mobilized uranium to produce the concentrations observed in 2010. Accordingly, it would have been necessary that significant downward flow occurred in the saturated zone as groundwater levels gradually declined upon passage of the flood wave in the river. Downward advective transport of the uranium would have been particularly important for detecting uranium at a concentration of 2.7 mg/L in well 0707, as the top of the 5-foot screened interval for this well is located 9.8 feet bgs, which is about 4.5 feet below the lowest water table levels observed at this well.

Several numerical simulations were performed with a cross-sectional model to assess the likelihood that contaminants mobilized from shallow floodplain sediments in June 2010 migrated as deep as the top of the screened interval in well 0707. The model was designed to account for groundwater flow and advective transport along a 2,100-foot section of the alluvial aquifer aligned with axis of the uranium plume as mapped over the past nine years. The model was constructed using 16 layers, a single row representing a flow path extending northwestward from the river to Rendezvous Road, and 105 columns with a uniform length of 20 feet. Temporally

variable, prescribed-head boundary conditions were invoked on the downgradient end of the model to represent changing river levels associated with flows observed at the U.S. Geological Survey gaging station on the river near Riverton during the 2010 flood event. A general head boundary condition was applied to the upgradient boundary to account for groundwater flow in the vicinity of Rendezvous Road, which during non-flood conditions comprises inflows from the northwest. Advective transport was simulated using particle tracking.

Several simulations were conducted with the cross-sectional model, each with a different combination of aquifer hydraulic conductivity and specific yield, with the intent of accounting for a variety of possible groundwater flow conditions. In all cases, the particle tracking indicated that the downward flow associated with a falling water table after the flood event was insufficient for driving leached contamination as deep as the top-of-screen elevation at well 0707. Thus the cross-sectional modeling suggested that, though some mobilization of unsaturated-zone contamination was likely, it was not large enough to produce the spike in uranium concentration observed at well 0707 in 2010.

Scenario 2 invokes the possibility that, during the period of overbank flows that occurred in June 2010, infiltration of surface water on the floodplain augmented downward flow in the aquifer to the extent that uranium was carried as deep as the top-of-screen elevation at well 0707.

Review of principles of contaminant transport associated with this scenario is helpful for assessing its viability. For example, for scenario 2 to be true, it is a requirement that the leaching of shallow floodplain sediments in June 2010 produced aqueous-phase concentrations of uranium that exceeded the 2.7 mg/L peak concentration observed at well 0707, and that such large concentrations were maintained over a several-day period. Otherwise, it would have been impossible for the shallow sediments to be the source of the uranium levels observed at the well. None of the laboratory leach-tests on soil samples produced a dissolved uranium concentration as large as 2.7 mg/L, which raises questions of the viability of this scenario.

The validity of scenario 2 was also analyzed with multiple simulations of two-dimensional groundwater flow and transport in a large area surrounding the former Riverton site over a time period that spanned the 2010 flood event. The numerical model used for this purpose was assigned a single layer intended to represent the entire saturated thickness of the alluvial aquifer, the Little Wind River comprised the southeast boundary of the simulation domain, and the flow portion of the model was automatically calibrated with the pilot-point methods incorporated in PEST software. In simulations aimed at accounting for uranium mass mobilization from shallow floodplain sediments, prescribed hydraulic heads along the model's downgradient border were varied over time to reflect changing river stages associated with river flows in 2010. Thus simulated groundwater levels rose and fell accordingly, due to pressure wave transmission. Mass loading of contamination to the alluvial aquifer was modeled in all cases by simulating recharge with a specified uranium concentration to floodplain areas in the vicinity of and upgradient of wells 0707 and 0789. The recharge represented infiltration of overbank surface water, and was only invoked over the time span when the river apparently overflowed its banks.

Several different combinations of recharge rate and prescribed concentrations were applied in the multiple simulations. One of the findings from the various model runs was that it was necessary to limit the assigned recharge rate so that computed groundwater levels remained within a range that did not exceed the estimated surface-water elevation during the overbank flood stage. In addition, it was necessary to use prescribed uranium concentration in the recharge on the order of

10 mg/L or greater to achieve simulated concentrations in the well 0707 and well 0789 areas that were of the same general magnitude as those measured after passage of the 2010 flood wave. These results suggest it is unlikely that this scenario was the cause of the uranium concentrations spikes observed in wells 0707 and 0789.

### **3.4 Groundwater Characterization**

#### **3.4.1 Summary of Method**

Groundwater samples were collected at all 103 borehole locations (Figure 26) according to the procedures specified in the Work Plan. Samples were collected after the Geoprobe rods were driven to 12 feet bgs or until rod refusal (see the Data Summary Report [DOE 2013a] for the locations where Geoprobe rods could not be driven to 12 feet), and 8 liters of water were purged from the rod. The 12-foot depth the Geoprobe rods were driven to approximated the average midpoint of the screened interval for the monitoring wells in the surficial aquifer that comprise the long-term monitoring network. Field measurements of pH, specific conductance, temperature, oxidation-reduction potential, total alkalinity, turbidity, and dissolved oxygen were made at each borehole, and samples were analyzed for the U.S. Nuclear Regulatory Commission (NRC)-approved contaminants of concern (manganese, molybdenum, sulfate, and uranium) (DOE 1998a), major cations (calcium, magnesium, potassium, and sodium), and an additional major anion (chloride). Samples were analyzed by ALS Laboratory Group in Fort Collins, Colorado, using standard EPA methods. Groundwater data were validated according to the "Standard Practice for Validation of Laboratory Data" in the *Environmental Procedures Catalog* (LMS/POL/S04325).

#### **3.4.2 Interpretation and Results**

##### **3.4.2.1 Comparability of Data**

An assessment of the comparability of the groundwater results obtained during the enhanced characterization to the results from the long-term monitoring network was conducted because sampling methodology was by necessity different for the temporary boreholes sampled during the enhanced characterization (DOE 2013a). The assessment was conducted by comparing (1) molybdenum, sulfate, and uranium results from monitoring well samples collected during the June 2012 routine sampling event with (2) results from the temporary boreholes collected in August 2012, as shown in Table 7. The temporary borehole closest to a monitoring well and within the same contour was used for the comparison.

As shown in Table 7, the mean percent difference (PD) ranged from -31.9 to 3.2 PD. This range of PD indicates good comparability of methods given the following considerations: (1) EPA guidance for acceptable precision for laboratory duplicates is 20 relative percent difference (LMS/POL/S04325), and one third of the comparisons met the laboratory criteria; (2) temporal variability between the June and August events; and (3) distance between the monitoring well and temporary borehole (up to 680 feet). Correlation between enhanced characterization results and long-term monitoring results was exceptional where uranium concentrations were high in monitoring wells 0707 and 0789.

Table 7. Comparison of June 2012 Results with August 2012 Results<sup>a</sup>

Monitoring Well Location	U June	U August	PD <sup>b</sup>	Mo June	Mo August	PD	SO <sub>4</sub> June	SO <sub>4</sub> August	PD
0707	1	1.1	-9.5	0.9	0.53	51.7	3100	2300	29.6
0716	0.3	0.22	30.8	0.13	0.11	16.7	460	440	4.4
0718	0.16	0.42	-89.7	0.068	0.21	-102.2	2600	2600	0.0
0720	0.0063	0.0028	76.9	0.0013	0.0058	-126.8	190	320	-51.0
0722R	0.51	0.18	95.7	0.13	0.15	-14.3	840	130	146.4
0729	0.0031	0.0096	-102.4	0.002	0.0046	-78.8	74	120	-47.4
0784	0.0028	0.0011	87.2	0.0099	0.018	-58.1	2300	2200	4.4
0788	0.053	0.029	58.5	0.022	0.02	9.5	1700	1200	34.5
0789	2.1	2.1	0.0	0.56	0.56	0.0	5900	3900	40.8
0824	0.0085	0.027	-104.2	0.0047	0.0057	-19.2	85	320	-116.0
0826	0.049	0.07	-35.3	0.02	0.027	-29.8	1800	2000	-10.5
		Mean	0.7		Mean	-31.9		Mean	3.2

<sup>a</sup> Units are in milligrams per liter (mg/L).

<sup>b</sup> Percent difference was calculated as  $[(a - b) + (a + b)/2] \times 100$ , where  $a$  = June concentration value from a monitoring well and  $b$  = August concentration value from a temporary borehole near the monitoring well.

### 3.4.2.2 General Water Chemistry

Major anion and cation data are displayed as a Piper diagram in Figure 34. Locations were divided into four areas of the aquifer—upgradient of the uranium plume, within the uranium plume, northeast of the uranium plume, and southwest of the uranium plume—and plotted on the diagram. General water chemistry varies spatially within the aquifer as shown in Figure 34:

- Upgradient locations (green) have no dominant cation type and are distributed between bicarbonate and sulfate types of water for anions.
- Locations within the uranium plume (red) tend to have no dominant cation type and are a sulfate type of water for anions.
- Locations northeast of the uranium plume (blue) are calcium type of water for cations and bicarbonate type of water for anions.
- Locations southwest of the uranium plume (black) are distributed between no dominant type and a sodium/potassium type of water for cations and are generally a sulfate type of water for anions.
- The difference between water types on each side of the uranium plume is likely due to the influence of sulfate in the discharge water from the sulfuric acid plant on the southwest side of the uranium plume that is recharging the surficial aquifer.

Groundwater summary statistics for all results are provided in Table 8, and a complete set of groundwater data collected during this characterization is provided in Appendix G.



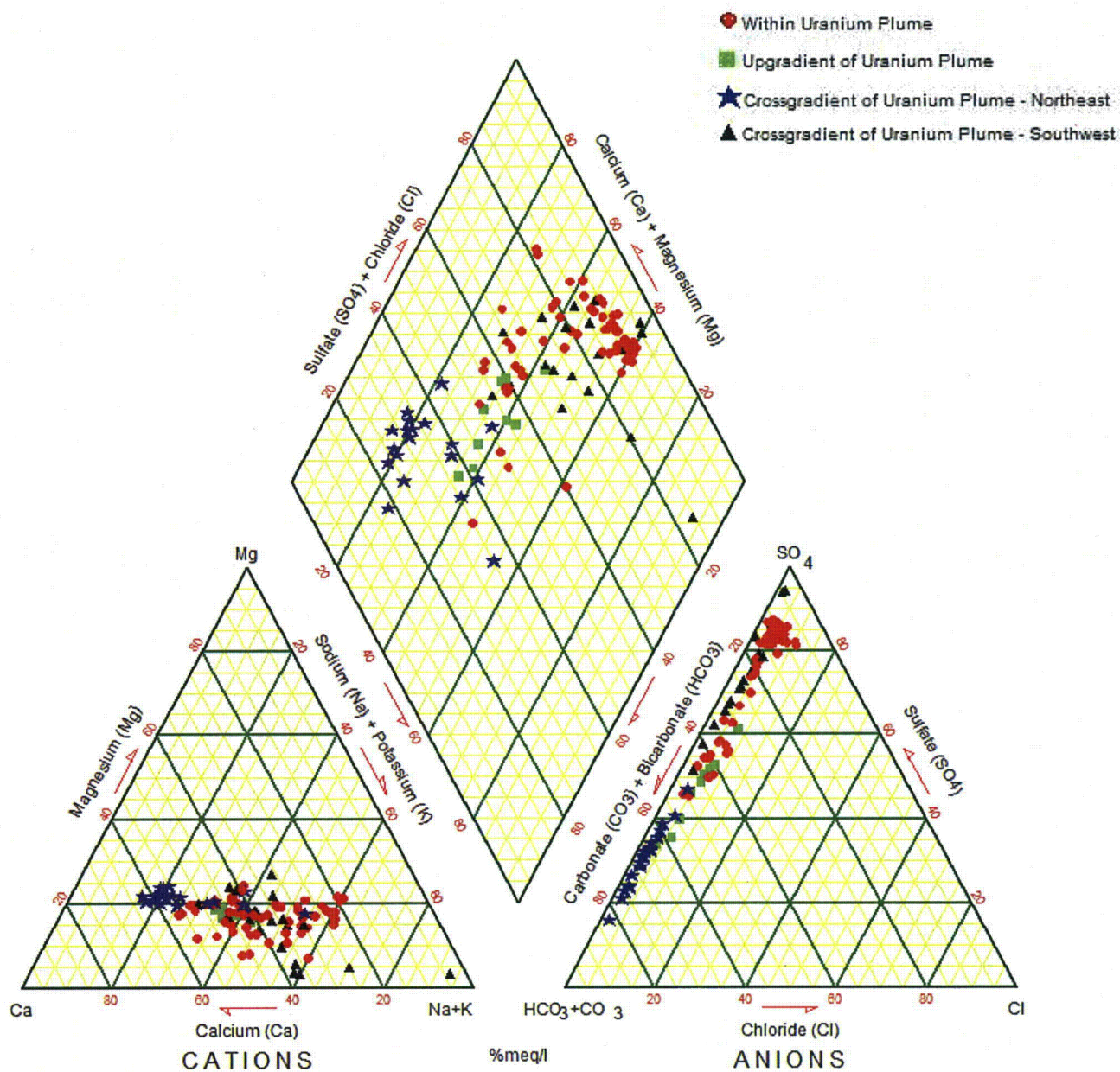


Figure 34. Piper Diagram of Major Anion and Cation Data

Table 8. Summary of Groundwater Results

Analyte	Benchmark <sup>a,b</sup>	Range <sup>b</sup>	Mean <sup>b</sup>	Area of Plume <sup>c</sup> (Acres)
Manganese	2.26	0.012–7.2	0.998	71
Molybdenum	0.1	0.004–1.1	0.165	182
Sulfate	400	39–5,900	1,431	465
Uranium	0.044	0.00081–2.1	0.277	323
Calcium	271	48–760	247	NA
Magnesium	25.5	7.7–390	76.4	NA
Potassium	4.1	2.6–28	8.8	NA
Sodium	167	16–2,000	429	NA
Chloride	73	3.4–570	72	NA

<sup>a</sup> Benchmark is either 40 CFR 192 MCL (molybdenum and uranium) or maximum background concentration (manganese, sulfate, calcium, magnesium, potassium, sodium, and chloride) (DOE 2012a).

<sup>b</sup> Units are in milligrams per liter (mg/L).

<sup>c</sup> Area of plume determined from outer contour of plume using Geographic Information System (GIS) software.

### 3.4.2.3 Manganese

Manganese concentrations in the surficial aquifer are relatively low, with the maximum concentration approximately four times the background concentration; therefore, manganese does not form a well-defined plume, as shown in Figure 37. Graduated symbol plots and box-and-whisker plots for all groundwater COPCs are shown in Figure 35 and Figure 36, respectively. Those figures show that higher manganese concentrations are skewed to the southwest and occur further upgradient than the molybdenum and uranium plumes.

### 3.4.2.4 Molybdenum

The molybdenum plume is narrow, well defined, and within the bounds of the long-term monitoring well network as shown in Figure 38. Figure 35 and Figure 36 confirm that elevated molybdenum concentrations occur within the narrow plume area and are evenly distributed along the axis of the plume. The current long-term monitoring network is adequate for monitoring molybdenum.

### 3.4.2.5 Sulfate

The sulfate plume (Figure 35, Figure 36, and Figure 39) is larger than the other plumes and is skewed upgradient and to the west, which is likely due to infiltration of water from the unlined ditch that carries discharge from the sulfuric acid plant. The outer boundary of the sulfate plume is defined as 400 mg/L, which is the maximum concentration observed in background wells (DOE 2012a). The ditch contained water with sulfate concentrations up to 2,000 milligrams per liter (mg/L) in 2012. Future work may be required to determine the contribution of sulfate to the surficial aquifer from the sulfuric acid plant versus the former mill; however, the high levels of sulfate in the centroid of the plume are likely mill-related.

### 3.4.2.6 Uranium

In general, the extent of the uranium plume (Figure 40) is similar to previous interpretations using monitoring data from the long-term monitoring program. The centroid of the plume is near the Little Wind River and located near monitoring well 0789, and the maximum uranium concentration found during this investigation is equal to the uranium concentration in monitoring well 0789.

However, two anomalous areas were revealed during the enhanced characterization. First, the uranium concentration measured in the sample collected at location T06-01 (furthest southwest location) was above the MCL in 40 CFR 192, which is anomalous for this area of the aquifer based on plume configurations and groundwater flow direction. This uranium concentration also stands out in Figure 35 and Figure 36. Although anomalous, this uranium concentration is considered valid as it correlates with the elevated sulfate concentration (1,200 mg/L) in T06-01 and elevated uranium concentration (above background) in samples collected from adjacent locations in the same transect (0.024 mg/L in T06-02, 0.02 mg/L in T06-03, and 0.029 mg/L in T06-04). Additional investigation work may be warranted in this area to determine the extent of uranium contamination.

Second, the uranium concentration at T03-08 (1.1 mg/L) on the south edge of the former tailings pile is higher than would be expected 23 years after the completion of surface remediation at the former mill site. This elevated concentration indicates more complex aquifer properties, geochemical controls, and/or additional sources/sinks, which enable recalcitrant uranium concentrations to remain upgradient of the main centroid of the plume. It is unknown if this is an isolated point, because other planned locations to the southwest on Transect 3 were not sampled due to owner access and cultural resource survey issues. Additional investigation work may be warranted in this area to determine the extent and possible causes of this high uranium concentration.

## 3.5 Site Conceptual Model

Limited empirical data indicates that surprises occur in 20 to 30 percent of conceptual models, with a surprise being defined as new data that renders the site conceptual model invalid (Bredehoeft 2005). The flood of the Little Wind River in 2010 due to rapid snowmelt and rainfall caused increases in dissolved contaminant concentrations in groundwater wells and provided a “surprise” related to the original site conceptual model as detailed in the Site Observational Work Plan (SOWP) (DOE 1998b). An update to the original model is needed. However, there continues to be aspects of the site that are not well understood, so the site conceptual model will continue to evolve as new data are collected (see Section 5.0, “Summary and Recommendations”) and as alternative site conceptual models are tested. This section presents the major aspects of the original site conceptual model presented in the SOWP and presents aspects of an evolving site conceptual model that have been discovered since 2010. A generalized schematic of the original conceptual model and a revised (and evolving) site conceptual model are presented in Figure 41.



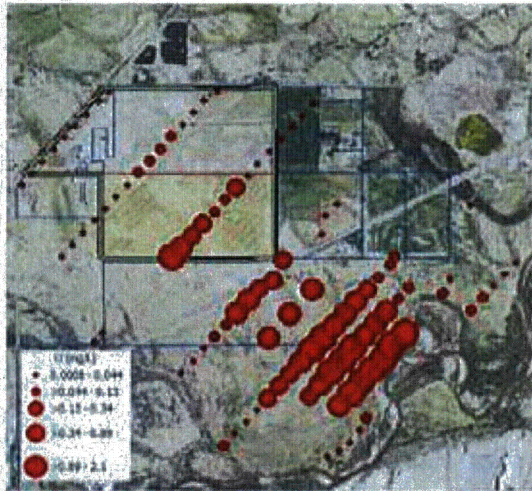


Figure a. Uranium (mg/L)

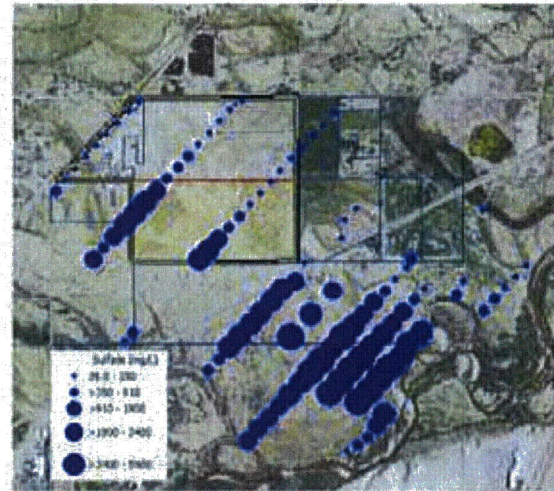


Figure b. Sulfate (mg/L)

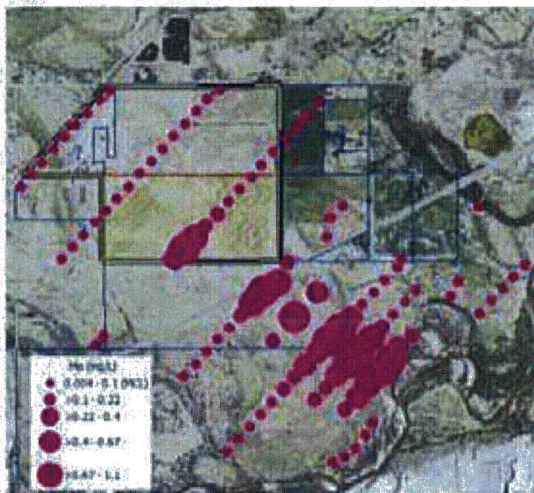


Figure c. Molybdenum (mg/L)

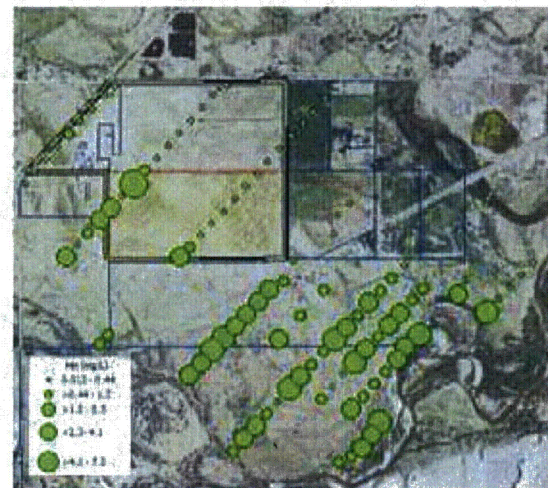


Figure d. Manganese (mg/L)

Symbol (magnitude) categories were established using the Natural Breaks (Jenks) classification method in ArcMap (version 10.0). Exceptions are the first category classes for U and Mo, which were modified slightly to correspond to MCLs (0.044 and 0.1 mg/L). (Initial Jenks-based ranges for these categories were 0.0008–0.037 and 0.004–0.075 mg/L for U and Mo, respectively.)

Figure 35. Graduated Symbol Plots of Manganese, Molybdenum, Sulfate, and Uranium in Groundwater: August 2012



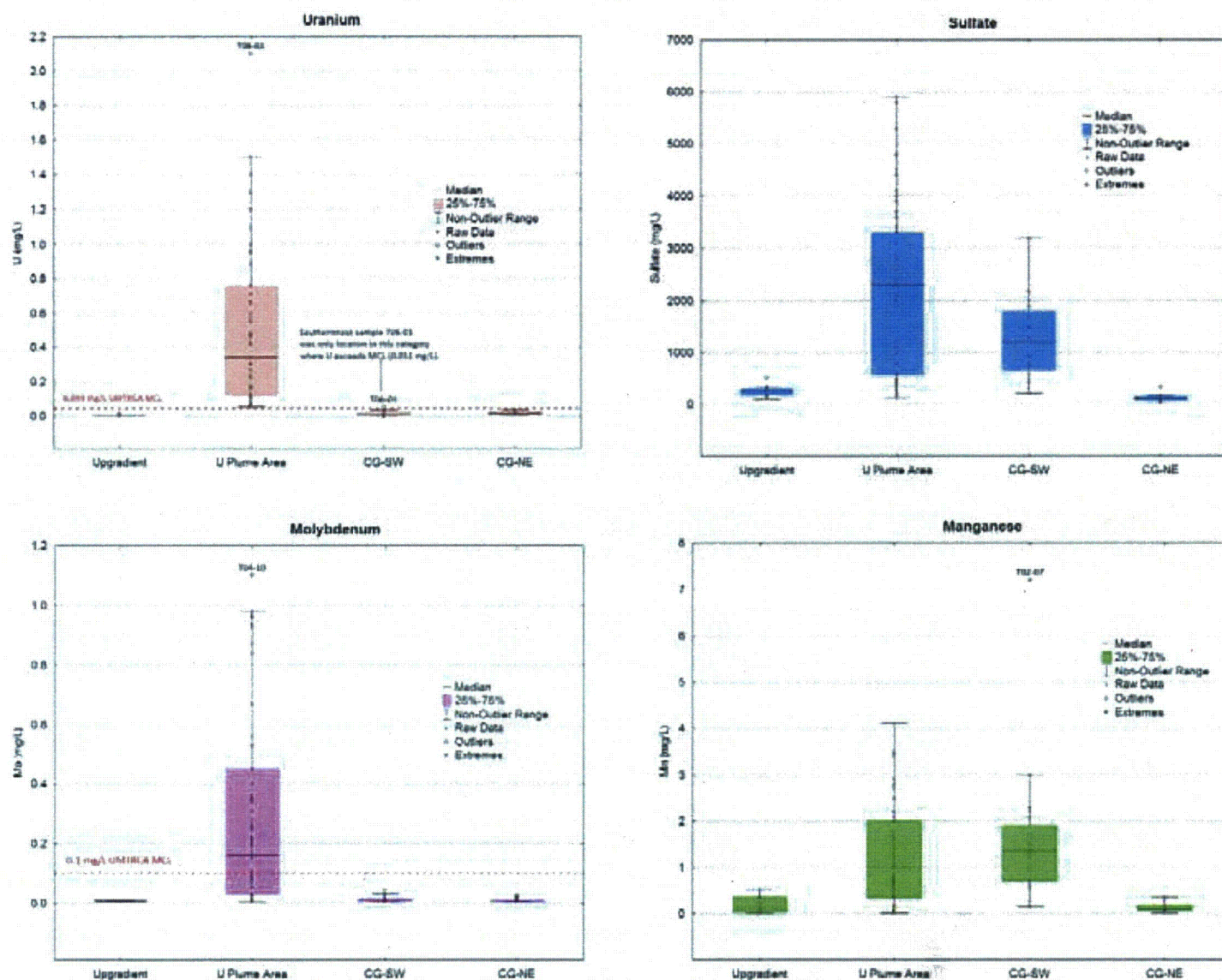


Figure 36. Box-and-Whisker Plots for Manganese, Molybdenum, Sulfate, and Uranium

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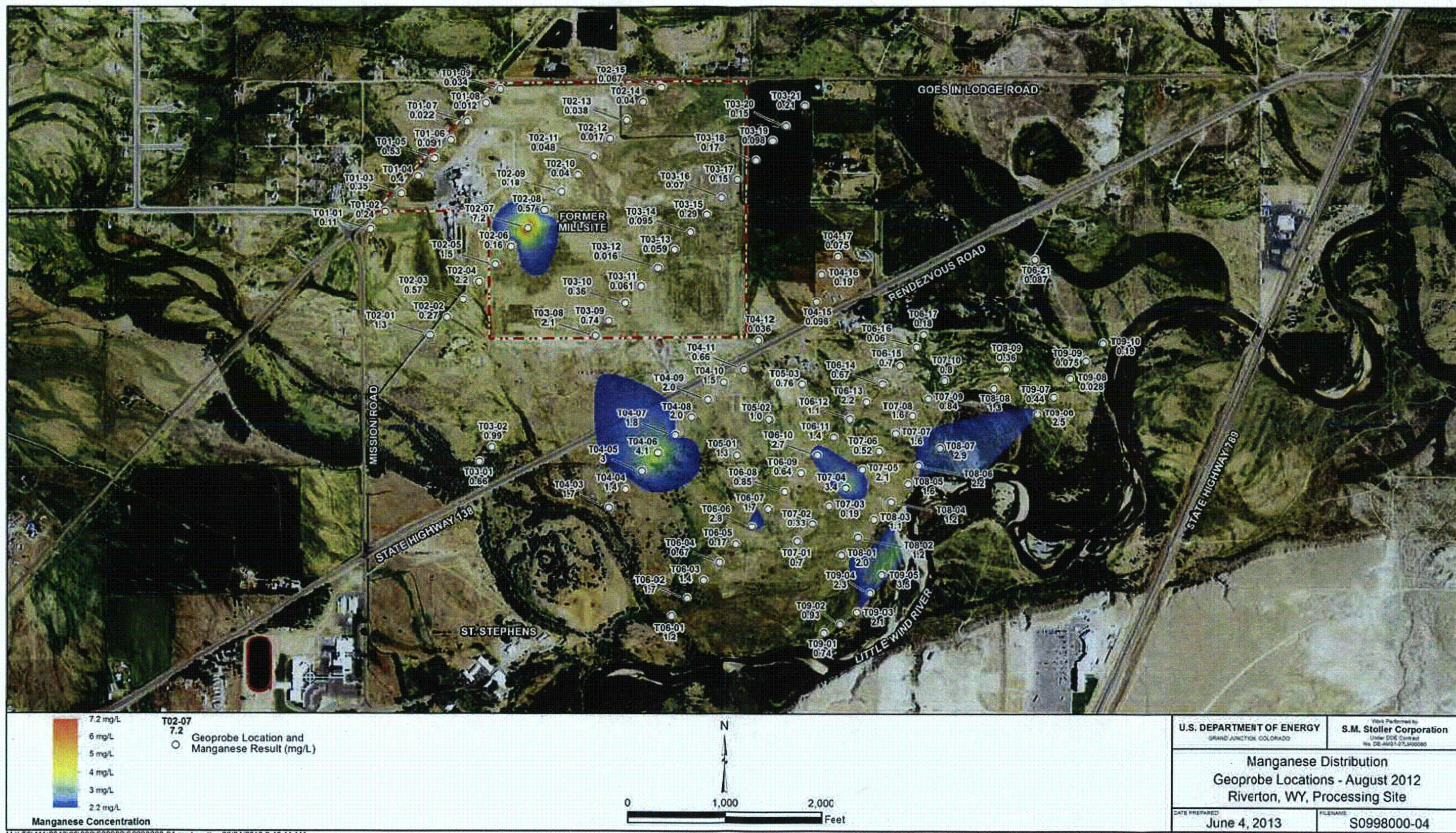


Figure 37. Distribution of Manganese in the Surficial Aquifer: August 2012 Enhanced Characterization



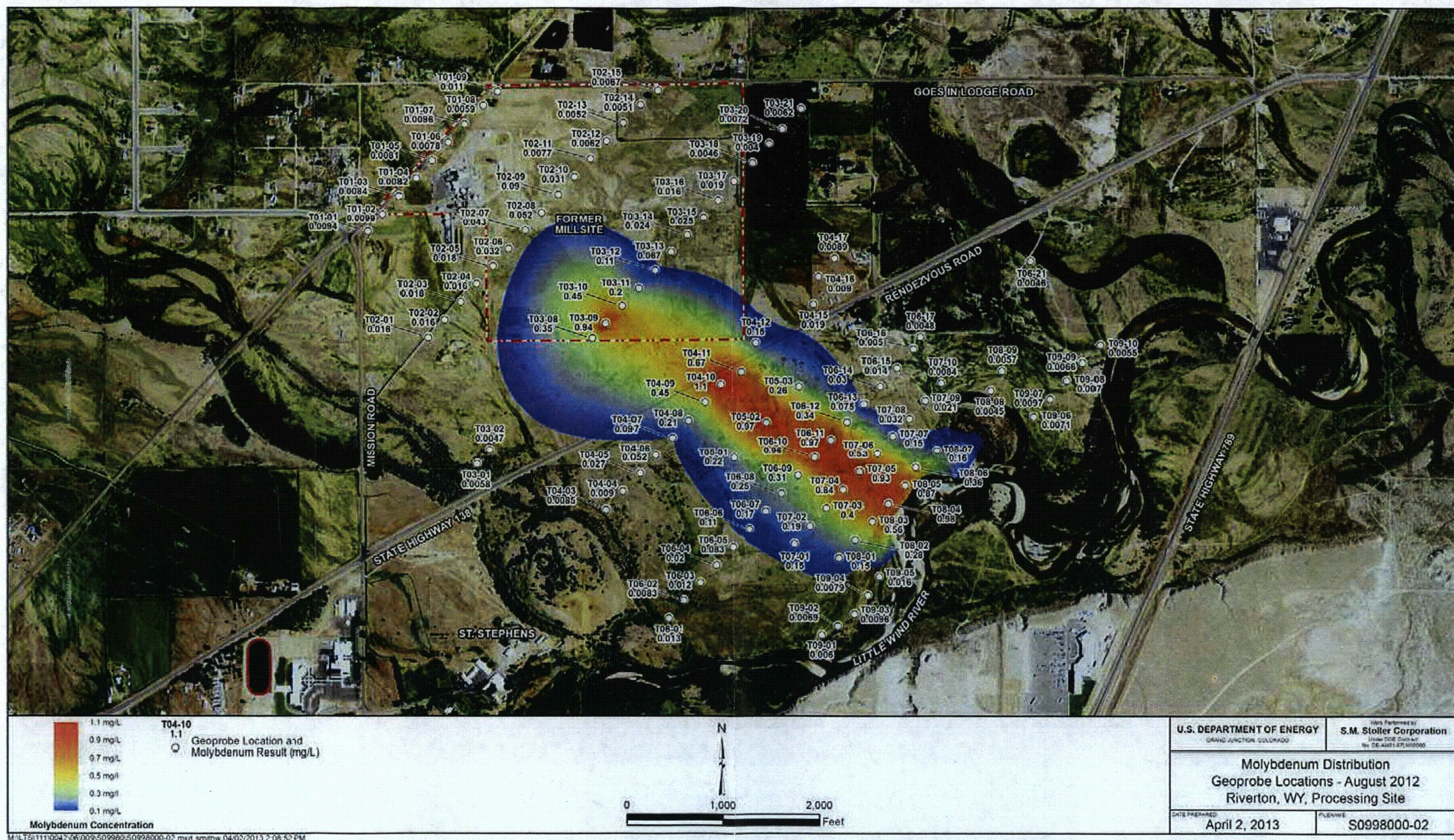


Figure 38. Distribution of Molybdenum in the Surficial Aquifer: August 2012 Enhanced Characterization



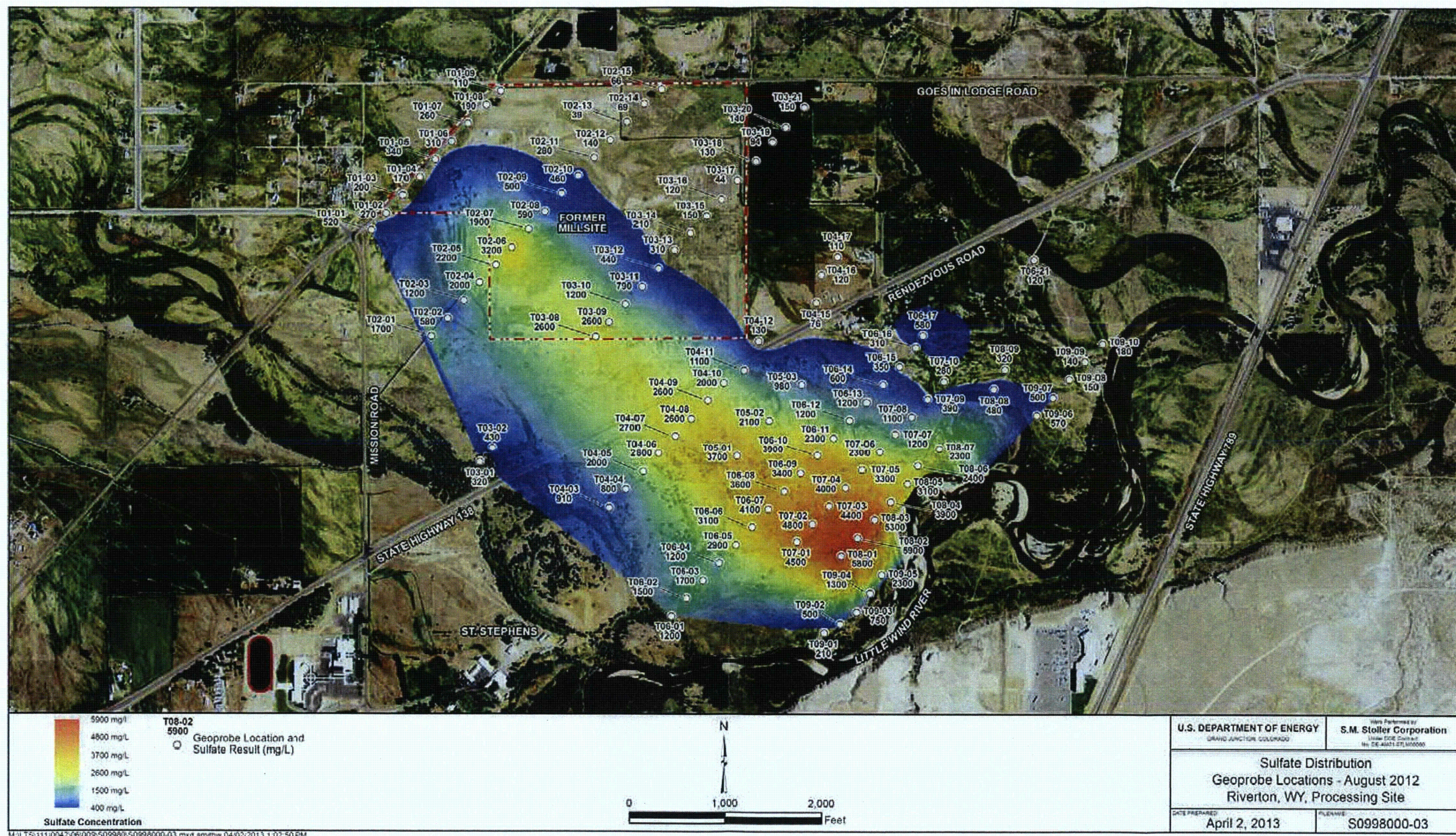


Figure 39. Distribution of Sulfate in the Surficial Aquifer: August 2012 Enhanced Characterization



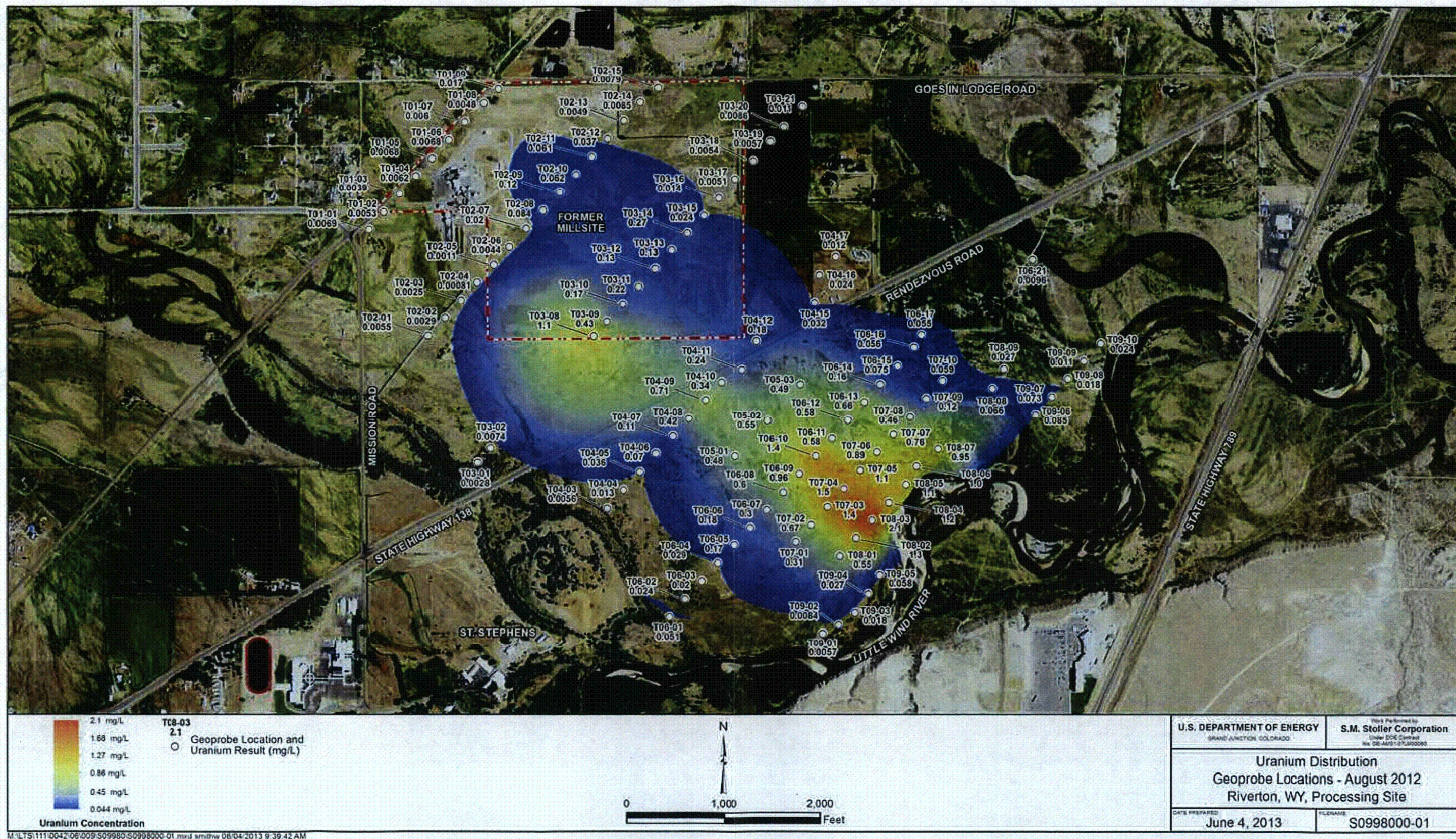


Figure 40. Distribution of Uranium in the Surficial Aquifer; August 2012 Enhanced Characterization



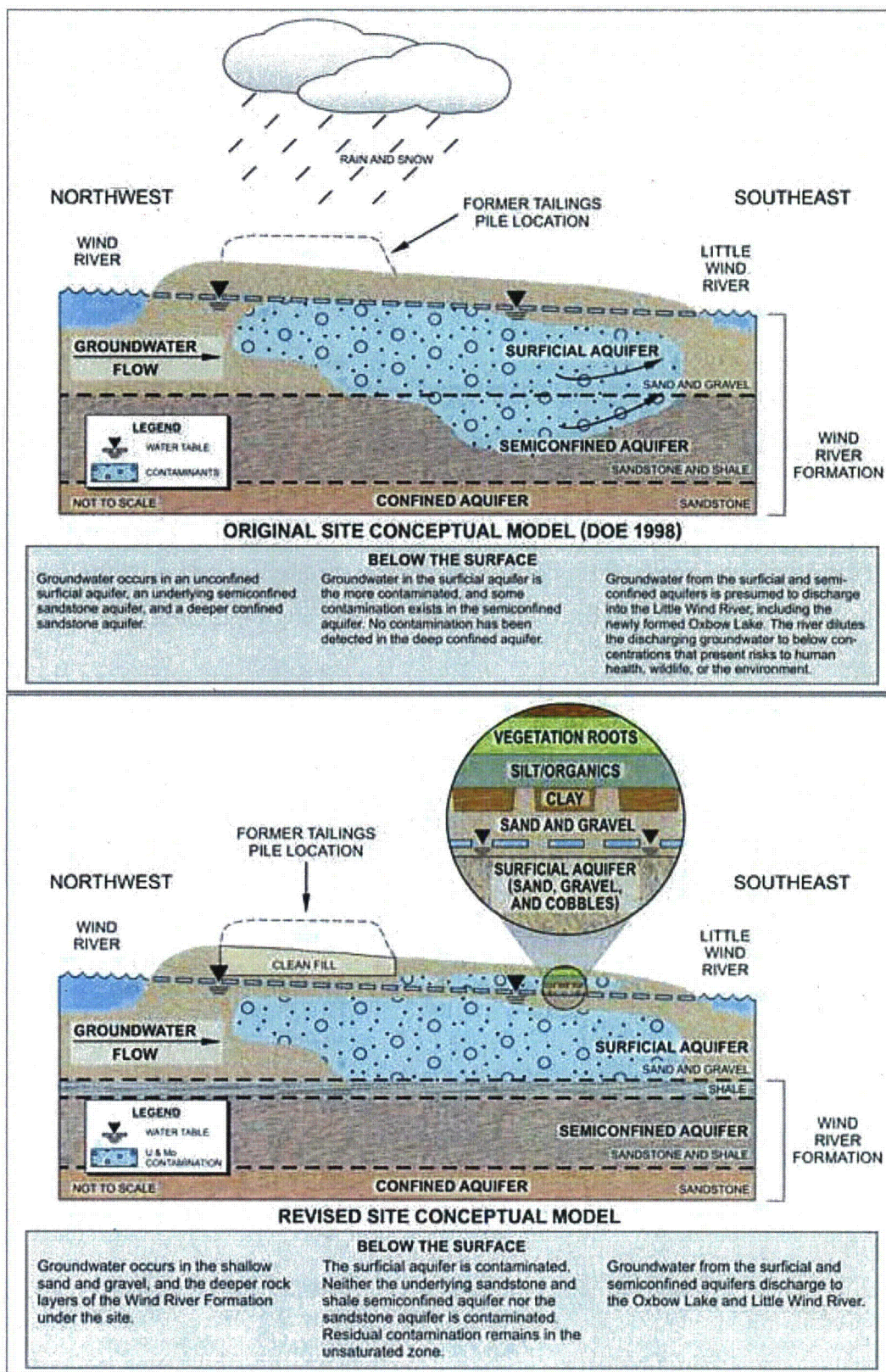


Figure 41. Original and Updated Site Conceptual Models

### **3.5.1 Original Site Conceptual Model**

This section lists the major aspects of the original site conceptual model as described in the SOWP.

#### **3.5.1.1 Original Contaminant Sources**

- Groundwater in the surficial aquifer was originally contaminated by downward migration of leachates from the former mill tailing pile as a result of transient drainage from tailings and from infiltration of precipitation on the pile.
- Tailings were not considered a source of continuing contamination, as tailings were excavated down to 4 feet bgs in 1989.
- The excavation also included contaminated surface soils outside the site boundary, which might have resulted from windblown tailings.
- All original sources of groundwater contamination were removed.

#### **3.5.1.2 Groundwater**

- Groundwater flows in the surficial aquifer from the Wind River to the Little Wind River. Flow direction can change seasonally – east-southeast in March and south in June.
- The surficial aquifer is unconfined with a geometric mean of saturated hydraulic conductivity of 125 ft/day.
- A discontinuous shale layer separates the sand and gravels of the surficial aquifer from the semiconfined aquifer; the two aquifers are hydrologically connected.
- The surficial aquifer and semiconfined aquifer discharge to the Little Wind River.
- Both the surficial and semiconfined aquifers have been impacted by site contaminants.
- The confined aquifer is hydrologically isolated from the other two aquifers and has not been impacted by site contaminants.

#### **3.5.1.3 Surface Water**

- The oxbow lake receives discharge of contaminated groundwater from the surficial aquifer.
  - The Little Wind River has not been impacted by site contaminants.
- Average river flow is 579 ft<sup>3</sup>/s and average groundwater discharge to the river 0.28 ft<sup>3</sup>/s.

#### **3.5.1.4 Groundwater Modeling/Natural Flushing Assessment**

- A GANDT probabilistic groundwater model was used to simulate groundwater flow and transport of uranium and molybdenum, assuming linear, equilibrium sorption (i.e., a  $K_d$  approach).
- All of the transport simulations were based on steady-state flow fields under non-flooding conditions.
- Hydraulic conductivity fields were created using geostatistical simulation techniques; hydraulic conductivities were allowed to vary from 1 to 180 ft/day.
- Modeling predicted that molybdenum and uranium levels would be below standards within 75 years of the 1998 starting time.

### 3.5.2 Revised Site Conceptual Model

This section lists new major concepts derived from additional data collection and evaluation since 2010. These new concepts represent changes and omissions from the original site conceptual model. These new concepts will form the basis for new investigations, data collection, and evaluations that will be used to test alternative site conceptual models and to refine and develop a new site conceptual model.

- Recalcitrant sources of contamination, or secondary sources, remain in the saturated and/or unsaturated zone of the alluvial aquifer.
- Spikes in groundwater contaminant concentrations occur as a result of hydraulic phenomena associated with river flood events that mobilize the secondary sources.
- Magnitudes of the concentration spikes in groundwater vary depending on the peak river flow associated with each high flow event, and may also be dependent on the duration of the event.
- Although the shale layer that separates the sand and gravels of the surficial aquifer from the semiconfined aquifer is discontinuous, there are enough fine-grained sediments in the upper portion of the Wind River Formation to prevent further downward migration of contaminants to more permeable strata within the Wind River Formation. Based on the presence of fine-grained sediments and low concentrations of uranium and molybdenum in the semiconfined aquifer monitoring wells, the semiconfined aquifer has not been impacted by site contaminants.
- Original groundwater modeling (which used steady-state flow fields and linear, equilibrium sorption, or the  $K_d$  approach) was too simplistic. It did not account for the effects of transient phenomena, such as changing flow conditions between seasons and the occasional mobilization of contaminants induced by river floods. The original modeling also did not account for additional transport processes that can greatly impact contaminant fate. Such transport phenomena include water chemistry-dependent desorption, rate-limited mass transfer from fine-grained to coarser-grained sediments, preferential flow zones, rate-limited mass transfer from intragrain porosity, and potential redox reactions in near-river areas. The site conceptual model will continue to evolve as these factors are evaluated.
- Hydraulic parameters used to estimate surficial aquifer properties have been updated based on additional site characterization (see Section 3.6 for details).

## 3.6 Groundwater Modeling

### 3.6.1 Modeling Approach

Observations in 2010 revealed that the existing numerical groundwater computer modeling did not account for the spikes in contaminant concentrations observed in the surficial aquifer groundwater after flooding of the Little Wind River. Consequently, the Work Plan specified that a new groundwater flow and transport model was needed to better simulate site conditions. Initial efforts were conducted to model flow and uranium source term in the unsaturated zone; however, the complexities of modeling the unsaturated zone, and the limited data for the unsaturated zone, made this impractical. Although this model did not account for additional

uranium source term, there are other aspects of the model that were updated from the original SOWP model. These modifications included:

- Extensive initial groundwater-concentration data generated from the enhanced characterization using the Geoprobe.
- An improved accounting for transient conditions and the influence of the Little Wind River flooding on the water levels in the surficial aquifer.
- An improved calculation of hydraulic conductivities using pilot points and PEST software (Doherty 1994).

This new model was intended to be one aspect of assessing the viability of the natural flushing compliance strategy, and so it should be viewed in light of other empirical evidence before a final decision is made. The new flow and transport model was intended to provide a conservative estimate of flushing time because of the following:

- The new model did not account for any additional source mobilized by flood events; therefore, actual flushing time will be longer than predicted by the new model because groundwater concentrations are known to increase after significant flood events.
- The lowest average  $K_d$  was selected from the laboratory tests conducted on soils that were similar to surficial aquifer materials. If a higher  $K_d$  was selected, flushing time predicted by the model would increase.

Four flow and transport models were developed. Three steady-state flow and transport models were developed to simulate unchanging flow conditions and to assist in development of a fourth model, which was a transient flow and transport model that is presented in this section. The transient model is considered more representative of the Riverton site because it can represent changing flow/stage of the Little Wind River over time.

### **3.6.2 Input Parameters, Assumptions, and Limitations**

Table 9 compares the input parameters in the new flow and transport model with the original GANDT model.

Using a groundwater flow and transport model has significant limitations, and the transport/forecasting aspect of the model should be viewed as a gross estimate, which should be interpreted only in conjunction with other lines of evidence. Data obtained from the enhanced characterization of the surficial aquifer revealed that concentrations of uranium were still high (1.1 mg/L) on the former mill site in 2012. This data, along with experience at similar sites and current literature (Zhu 2003), suggests that groundwater modeling using a linear, equilibrium sorption or  $K_d$  approach is too simplistic and does not account for fine-grained sediments and reducing zones (acting as variable sources/sinks in the aquifer) that make transport of contaminants erratic and unpredictable.

Table 9. Groundwater Model Inputs

Parameter	Units	New MODFLOW Model		Original GANDT Model	
		Value	Source	Value	Source
Hydraulic Conductivity	ft/day	6 to 433	Pilot points & PEST	57	DOE 1995, SNL 1996, and model trials
Recharge	ft/day	0.00016	Lasse 1998	0.0002	Expert judgment and general literature
Porosity	Decimal fraction	0.3	Lasse 1998	0.3	DOE 1995 and general literature
Dispersivity	ft	500	Expert judgment and general literature	160–230	Expert judgment and general literature
K <sub>d</sub>	mL/g	1.04	Laboratory soil testing	0.1–0.2	DOE 1993 and general literature
Bulk Density	g/cm <sup>3</sup>	2.5149	Lasse 1998	1.8	General literature
Initial Uranium Concentration	µg/L	0.81 to 2,100	August 2012 enhanced characterization	Not reported	DOE 1995
Background Uranium Concentration	µg/L	5	Mean from background well data	Not reported	DOE 1995

µg/L = micrograms per liter

### 3.6.3 Transient Flow Model

The new groundwater flow model is a single-layer, transient flow that was developed using MODFLOW 2000. Groundwater Vistas (GV) was used in conjunction with MODFLOW 2000. GV is a groundwater modeling environment for Microsoft Windows that couples a powerful model design system with comprehensive graphical analysis tools. GV is a model-independent graphical design system that can be used with MODFLOW and other similar models.

#### 3.6.3.1 Model Calibration

This groundwater flow model was calibrated using continuous water level data from 2005 and 2009 obtained from a transducer installed in monitoring well 0707 along with water level data obtained from the monitoring well network during routine sampling events in 2004, 2007, 2009, 2010, and 2011. Model calibration for transducer data is illustrated in Figure 42 and Figure 43 by comparing actual water elevations in monitoring well 0707 with simulated water levels generated by the groundwater model in 2005 and 2009, respectively. Calibration statistics were generated by looking at the difference (residual) between the modeled versus actual water level at 551 targets, which is a substantial number of targets (Table 10). A good “rule of thumb” for model calibration is a sum of squares of residuals per target of 1.0 or less. As shown in Table 10, the sum of squares per target is 0.31, which indicates good model calibration.



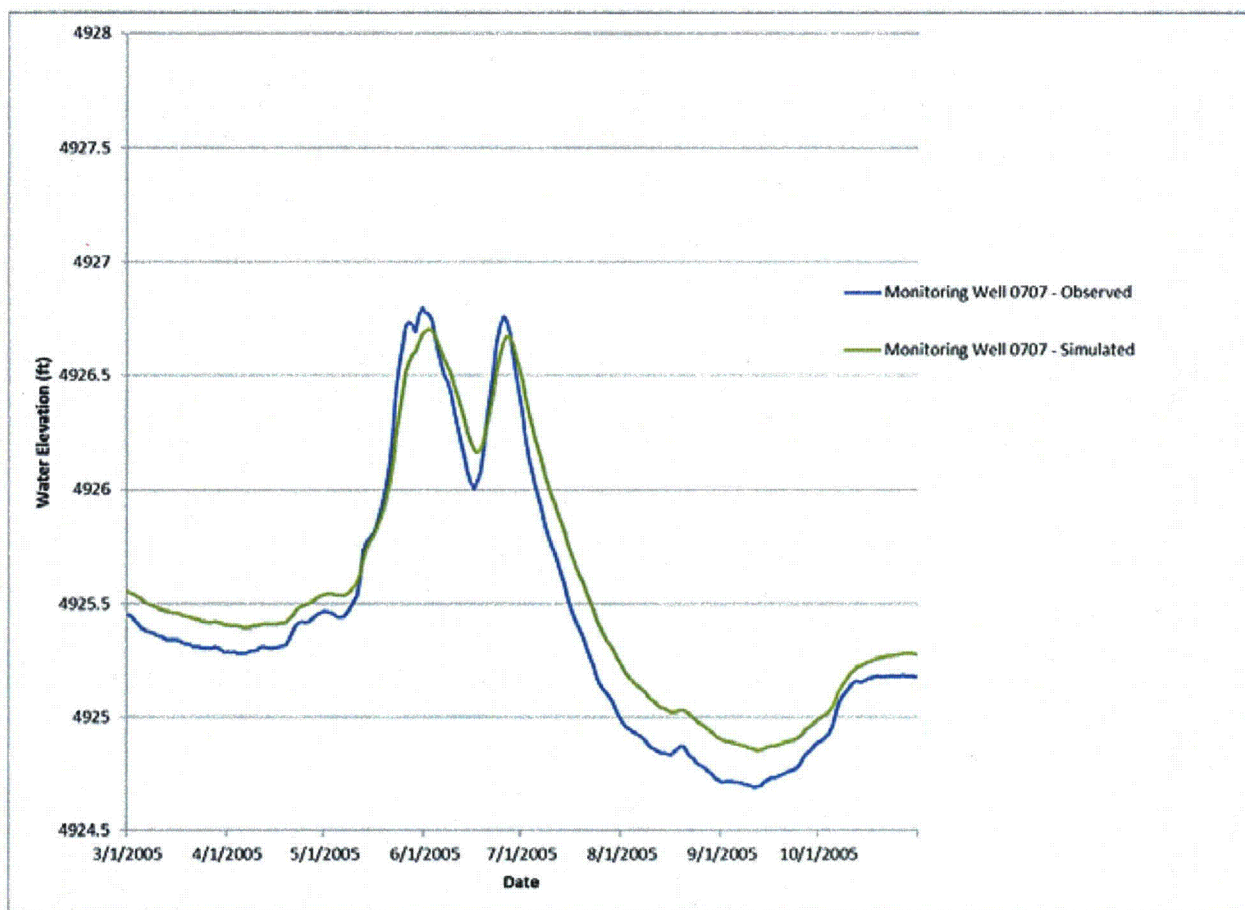


Figure 42. 2005 Water Levels versus Model Simulation: Well 0707



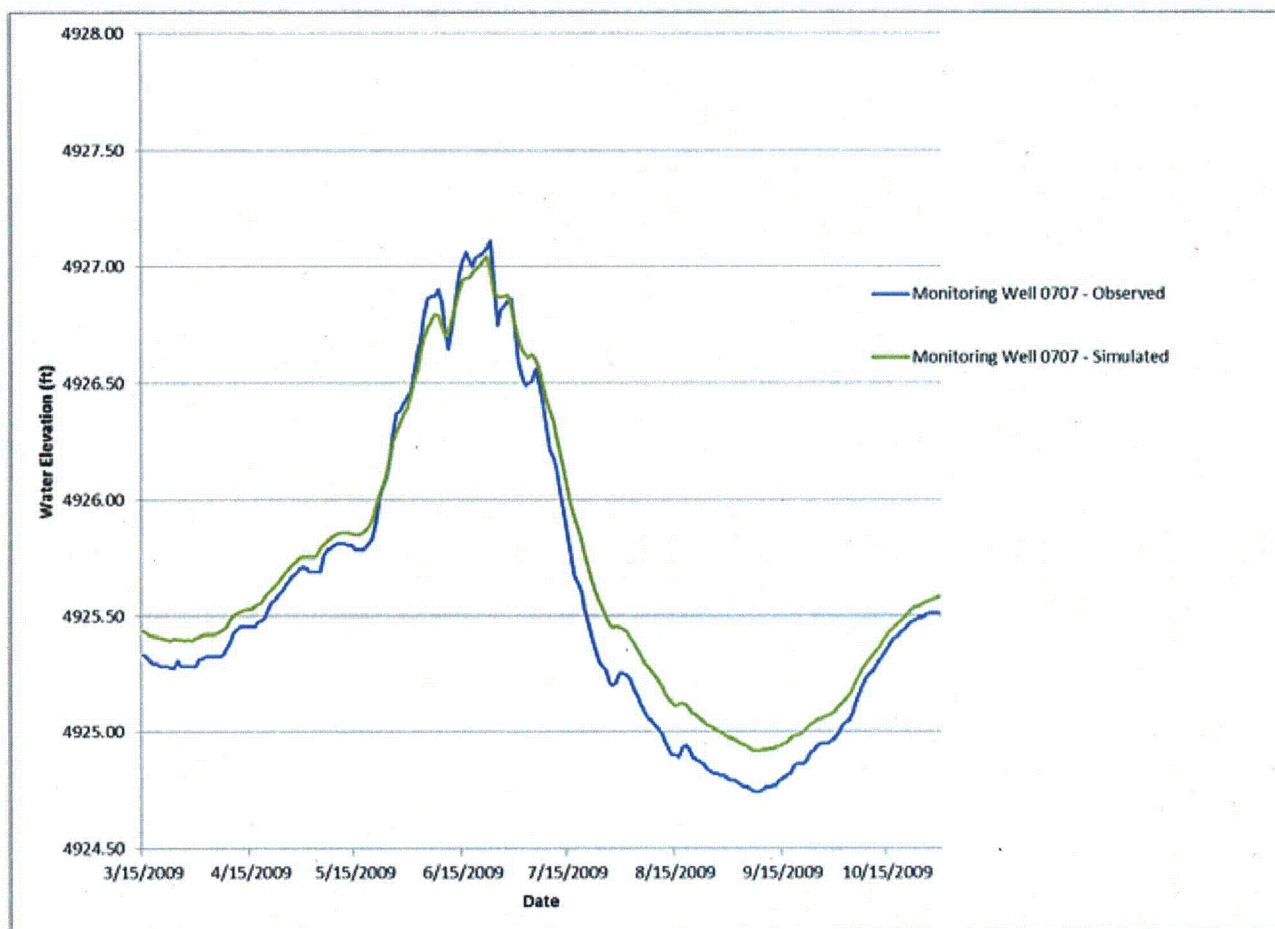


Figure 43. 2009 Water Levels versus Model Simulation: Well 0707

Table 10. Groundwater Model Calibration Statistics

Calibration Statistic	Value
Residual Mean	-0.087
Absolute Residual Mean	0.254
Residual Standard Deviation	0.553
Sum of Squares	172.7
Number of Targets	551
Sum of Squares per Target	0.31
RMS Error	0.560
Minimum Residual	-2.67
Maximum Residual	4.31
Range in Observations	24.15
Scaled Residual Standard Deviation	0.023
Scaled Absolute Residual Mean	0.011
Scaled RMS Error	0.023
Scaled Residual Mean	-0.004

Pilot points can be used for several parameters within GV in calibrating a model, including horizontal hydraulic conductivity ( $K_x$ ), vertical hydraulic conductivity ( $K_z$ ) specific storage ( $S_s$ ) specific yield ( $S_y$ ) recharge, and porosity. Sensitivity analysis indicated that the model was sensitive only to hydraulic conductivity, so pilot points were used for horizontal hydraulic conductivity to aid in the calibration of the Riverton model.

In conventional model calibration, the calibration process typically involves assigning one overall hydraulic conductivity value (or a separate hydraulic conductivity value to each hydraulic conductivity zone, if there are multiple zones), and adjusting this parameter (these parameters) until the fit between model-predicted and field-observation values is as good as possible. If the fit obtained on the basis of existing zones is not acceptable, then extra zones could be added into the model domain at locations where the modeler felt that they would “do the most good,” which is arbitrary. This process would continue until the fit between model predicted and observed values are acceptable. There are a number of shortcomings associated with this approach, which include:

- The process is labor intensive and slow.
- Often there is no geological mapping to provide guidance on where to put additional zones, which makes the process subjective and non-unique.
- Characterization of heterogeneity by zones of piecewise uniformity is not consistent with the nature of geological material, so that any zonation that is finally decided upon is defensible only on the basis that it is better to employ a zonation scheme than to ignore heterogeneity altogether. In addition, piecewise uniformity as a method of characterizing heterogeneity lacks the flexibility required to explore the effects of small-scale variability on model predictive uncertainty.

These problems can be overcome using pilot points and PEST software. PEST is a model-independent calibration tool from Watermark Computing. PEST uses nonlinear least-squares techniques to calibrate virtually any type of model. Special software is included with GV to interface PEST with all models supported by GV.

In the transient flow model (i.e., the new model), the distribution of hydraulic conductivity within the model domain was described by a set of pilot points. Pilot points were located in the model domain, and PEST was used to estimate the hydraulic conductivity of the aquifer at each point. These “point hydraulic conductivities” are then spatially interpolated to all the active cells within the model domain using kriging. In estimating hydraulic conductivity values at pilot points, PEST effectively assigns parameter values to the whole model domain.

A total of 91 pilot points were introduced into the model domain. Pilot points are associated with different site activities (pumping and slug tests) or model features (calibration targets), as shown in the Table 11. Each pilot point is assigned an initial value and a range to restrict hydraulic conductivity to reasonable values. The initial value and range for the pumping and slug tests vary by location and are displayed in Table 12. The hydraulic conductivity field generated using pilot points and PEST is shown in Figure 44.



Table 11. Pilot Points Summary

Parameter	Number of Pilot Points	Description	Initial Value and Range (ft/day)
Pump Tests	3	Pump test locations	Varies (see Table 12)
Slug Tests	5	Slug test locations	Varies (see Table 12)
Calibration Targets	9	Target locations	125.0, 100.0–400.0
Target Triangle	46	Center of each calibration target triangle	125.0, 100.0–400.0
Filler	28	Placed in cells that do not have pilot points within 10 cells	125.0, 100.0–400.0

Table 12. Pilot Points Details

Location	Pilot Point Type	Initial Value (ft/day)	Range (ft/day)
0100	Pumping	104.0	101.0–400.0
0737	Pumping	158.0	155.0–400.0
0738	Pumping	119.0	116.0–400.0
0724	Slug	5.9	4.9–400.0
0728	Slug	16.9	11.9–400.0
0729	Slug	5.4	4.4–400.0
0783	Slug	128.1	74.1–400.0
0788	Slug	7.4	12.4–400.0

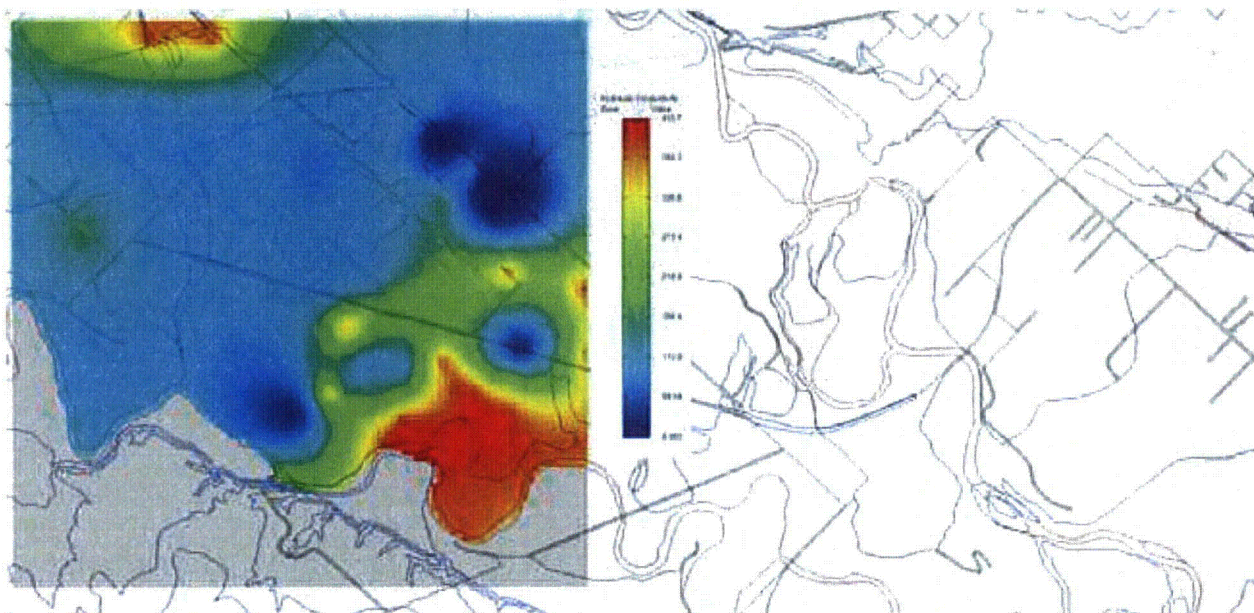


Figure 44. Hydraulic Conductivity Field Calculated Using Pilot Points and PEST



A statistical analysis of annual peak river flows in the Little Wind River was performed to develop perspective regarding the frequency with which the river could be expected to flood and help release contamination. The analysis, based on a record of annual peak flows extending from 1941 to 2011, revealed that the peak river discharge in 2010 (13,300 cfs) was likely to be equaled or exceeded once every 75 years (i.e., a 75-year return period). In addition, analysis of smaller flood events, such as the flood in June of 1965 (peak discharge of 9,550 cfs) had a return period for that peak flow of 15 years. These return periods of 15 and 75 years were used in the new transient flow model. In this model, the typical or average flow years were combined with a higher flow that occurs approximately every 15 years and an extreme flood flow that occurs approximately every 75 years. Constant head (representing river elevation) varies, based on typical or average flow, the 15-year flood event flow, and the 75-year flood event flow. The setup and summary of stress periods using these flood return periods are shown in Table 13 and Table 14, respectively.

*Table 13. Stress-Period Setup for the Transient Flow Model*

Year Type	Number of Stress-Periods	Stress-Period Length (days)	Cumulative Length (days)
Typical Year	1	90	90
	14	10	230
	1	135	365
15-Year Flood Event	1	125	125
	16	5	205
	1	160	365
75-Year Flood Event	1	125	125
	17	5	210
	1	155	365

*Table 14. Transient Flow Model Stress-Period Summary*

Flow Type	Number of Years	Date Range	Number of Stress-Periods per Year	Beginning Stress-Period	Ending Stress-Period
Typical	13	2012-2024	16	1	208
15 Yr. Flood	1	2025	18	209	226
Typical	14	2026-2039	16	227	450
15 Yr. Flood	1	2040	18	451	468
Typical	14	2041-2054	16	469	692
15 Yr. Flood	1	2055	18	693	710
Typical	14	2056-2069	16	711	934
15 Yr. Flood	1	2070	18	935	952
Typical	14	2071-2084	16	953	1176
75 Yr. Flood	1	2085	19	1177	1195
Typical	14	2086- 2099	16	1196	1419
15 Yr. Flood	1	2100	18	1420	1437
Typical	14	2101-2114	16	1438	1661
15 Yr. Flood	1	2115	18	1662	1679
Typical	14	2116-2129	16	1680	1903
<b>Total</b>	<b>118 Total</b>				



### 3.6.4 Transport Modeling and Forecasting

Transport simulations were conducted using MT3DMS software. Results of the new flow and transport model are presented below. This model was run for 118 years starting in 2012. Results indicate that the location of higher concentration is further east (downgradient) with increased river elevations during flood events. With higher river elevations, the gradient from the processing area toward the river in the vicinity of well 0707 is decreased. The flow direction likely shifts more to the southeast during 15-year flood and 75-year flood events, causing spreading of contamination in this direction. The change is clearly evident during a 75-year flood event. The initial concentration and transport simulations for 50-year and 100-year time frames are shown in Figure 45, Figure 46, and Figure 47, respectively. As shown in those figures, uranium concentrations are estimated to be above the standard after 100 years (in 2112). That will be 114 years since the Groundwater Compliance Action Plan (GCAP) was finalized in 1998. The GCAP predicted that natural flushing and other natural attenuation processes would reduce contaminant concentrations to MCL or background levels by the year 2098.



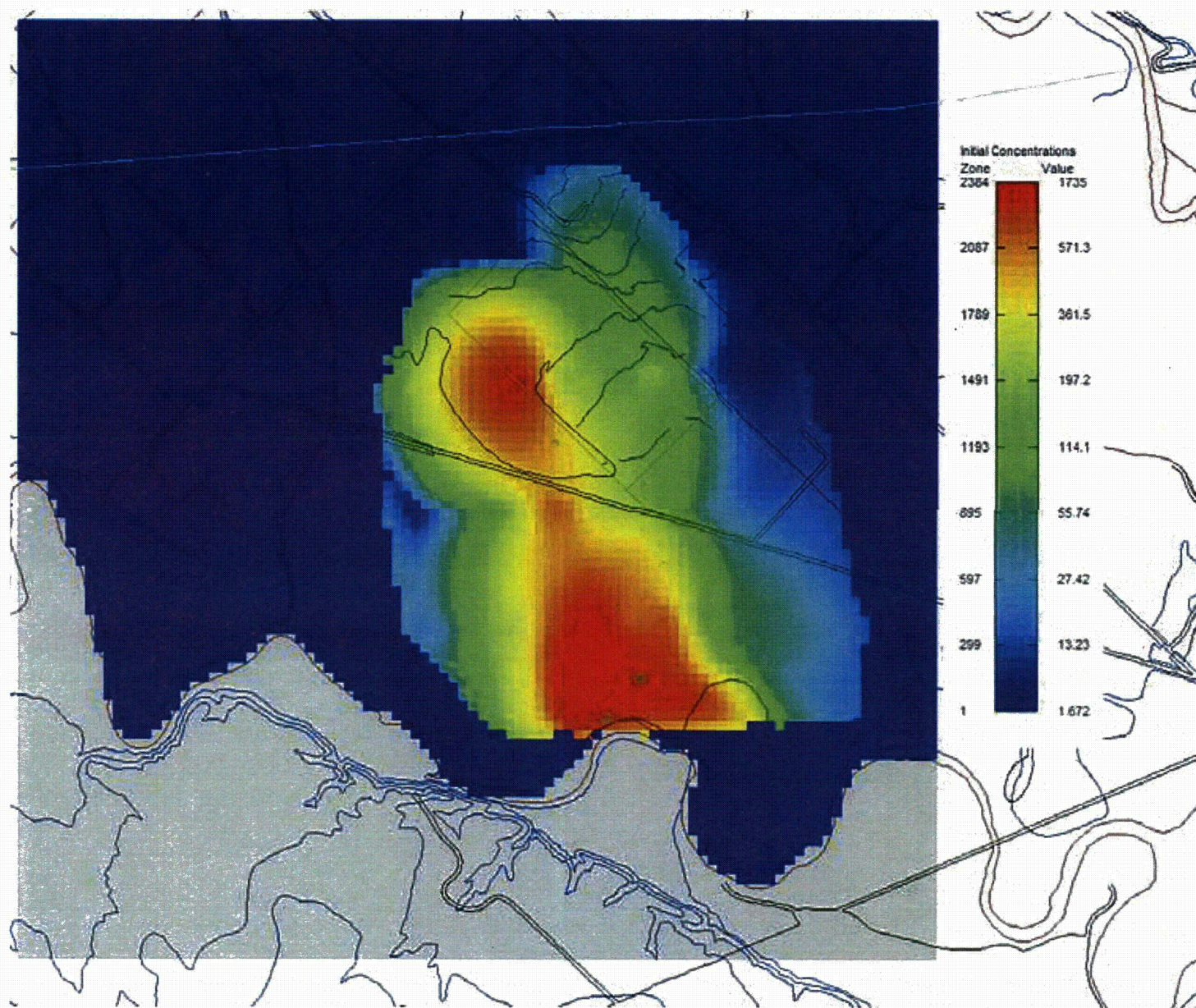


Figure 45. Initial Uranium Concentrations ( $\mu\text{g/L}$ ) in the Surficial Aquifer from the Enhanced Characterization – August 2012



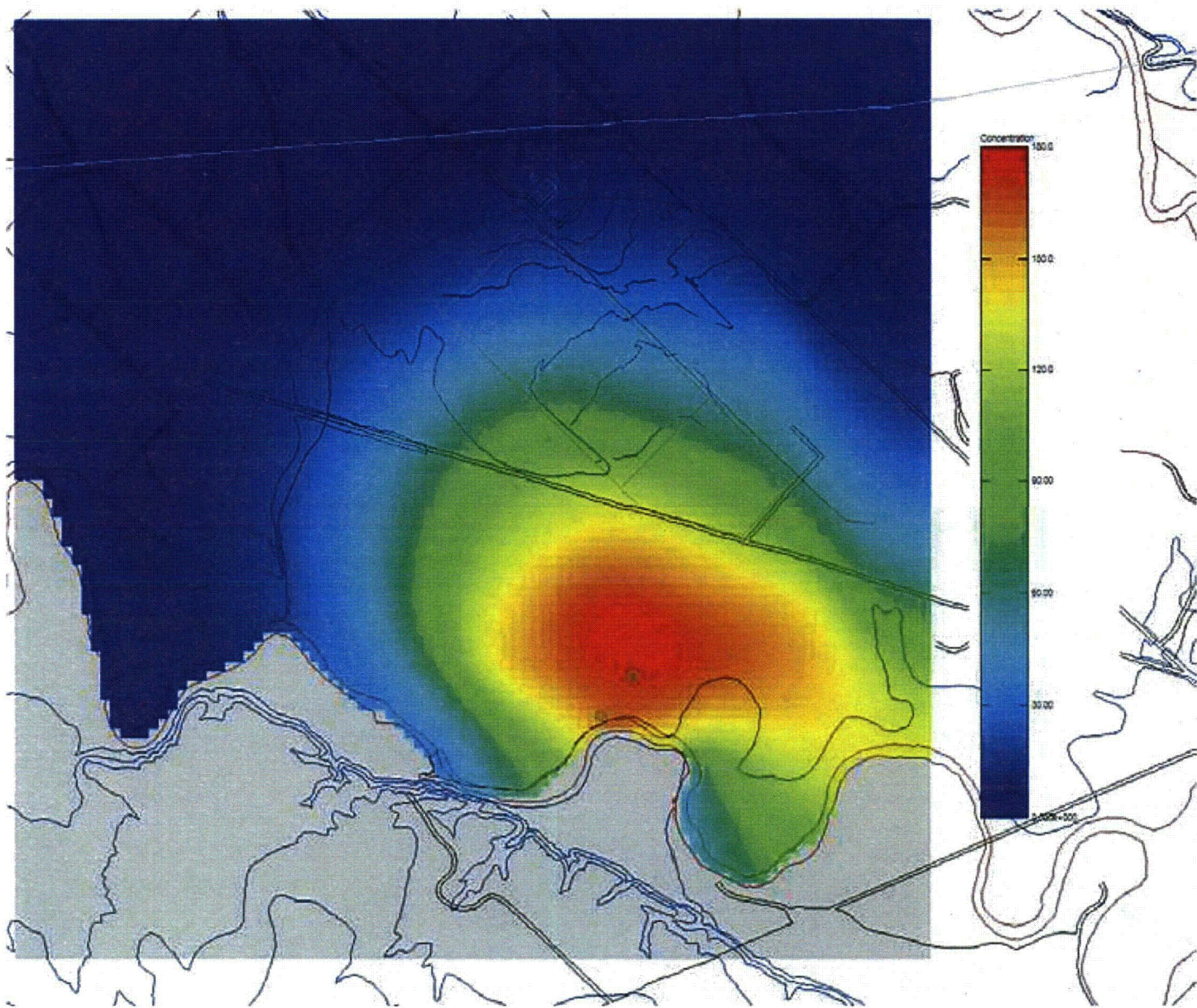


Figure 46. Simulated Uranium Concentrations ( $\mu\text{g/L}$ ) after 50 Years (i.e., in 2062)



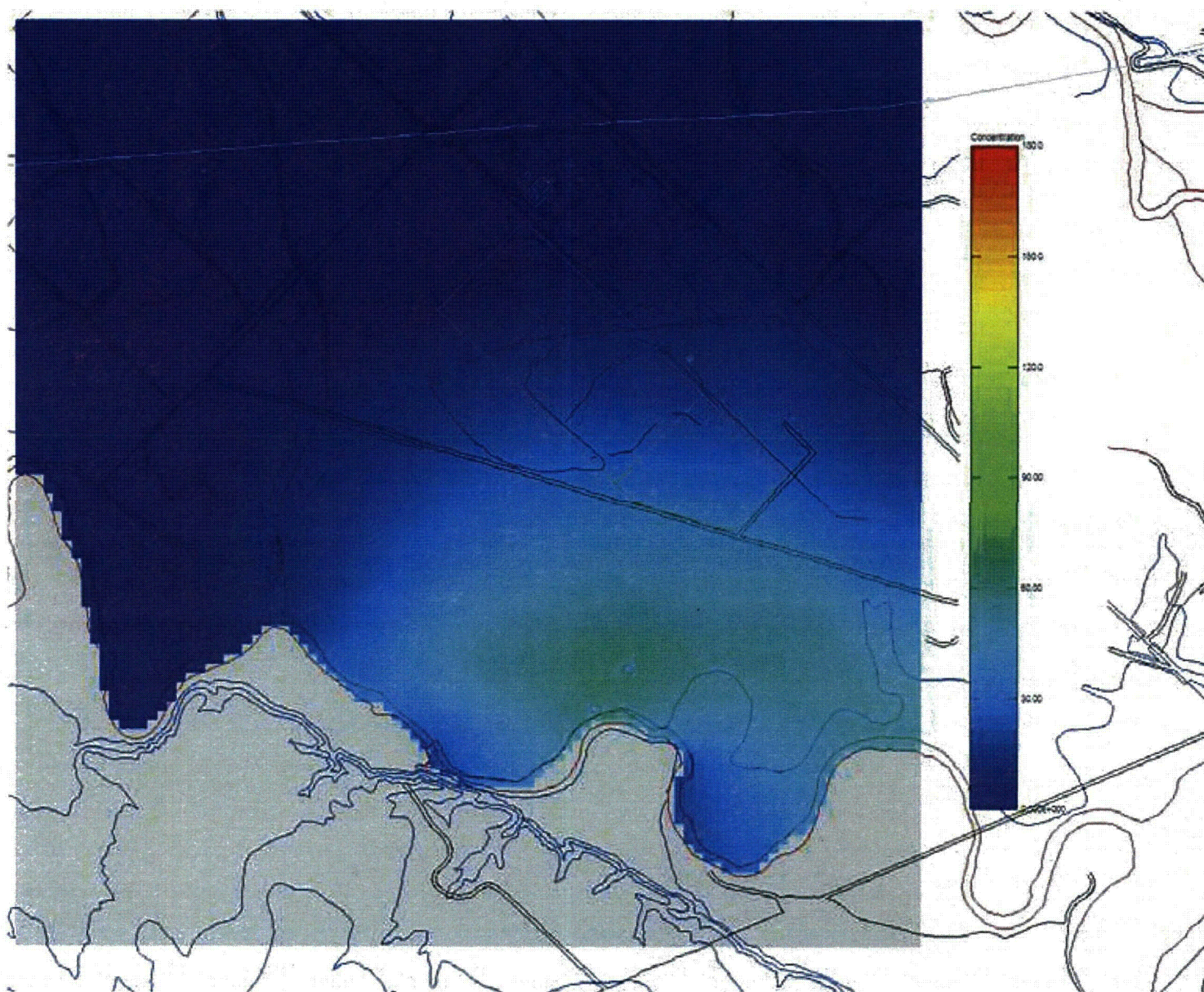


Figure 47. Simulated Uranium Concentrations ( $\mu\text{g/L}$ ) after 100 Years (i.e., in 2112)



## 4.0 Compliance Strategy Assessment

After surface remediation was completed, groundwater numerical modeling in 1998 predicted that the alluvial aquifer will naturally flush contaminants to levels below applicable standards within the 100-year regulatory time frame. This modeling formed the basis for the natural flushing strategy that was approved in the *Final Ground Water Compliance Action Plan for the Riverton, Wyoming, Title I UMTRA Project Site* (DOE 1998a) in 1998. In previous years, the progress of natural flushing was assessed using three tools: comparison to hydrogeologic modeling predictions, trend analysis, and curve matching/interpolation techniques applied to temporal plots of contaminant concentrations at individual locations. These techniques were based on a site conceptual model of gradually declining contaminant concentrations after surface remediation of source material on the former mill site. Prior to 2010, these techniques indicated that natural flushing of the surficial aquifer was progressing toward applicable standards.

However, based on observations made in 2010 in context with historical data, the site conceptual model and groundwater computer modeling were too simplistic to account for the spikes in contaminant concentrations in the surficial aquifer groundwater. Spikes in contaminant concentrations are attributed to flooding of the Little Wind River in June 2010, which mobilized contaminants into the saturated zone of the surficial aquifer. Cross correlation of flood events in the Little Wind River with monitoring data reveal that uranium concentrations spiked in monitoring well 0707 in 1991, 1995, and 2010, which followed floods of Little Wind River (Figure 48).

Although the 2010 flood of the Little Wind River caused significant spikes in contaminant concentrations in the surficial aquifer, contaminant concentrations continue to decline and are generally approaching pre-flood levels, as shown in Table 15. Figure 49 shows the average uranium concentration in surficial aquifer wells with a long history that have always been above the MCL (0707, 0716, 0718, and 0722/0722R). As shown in this Figure, the average uranium concentration in these wells was below pre-flood levels in 2012. These data indicate that the effects of 2010 flood are relatively short-lived in context of the 100-year regulatory time frame.

Table 15. Comparison of Pre-Flood, 2010 Flood, and 2012 Results

Well	Molybdenum <sup>a</sup>			Uranium <sup>a</sup>			Sulfate <sup>a</sup>		
	Pre-Flood <sup>b</sup>	2010 Flood <sup>c</sup>	2012 <sup>d</sup>	Pre-Flood	Post-Flood	2012	Pre-Flood	2010 Flood	2012
0707	0.68	1.6	0.85	0.84	2.7	0.85	1900	7000	3000
0788	0.024	0.023	0.022	0.034	0.1	0.048	630	4500	1500
0789	0.56	0.51	0.66	1.5	2.5	2	3900	9400	5300
0826	0.023	0.046	0.021	0.041	0.08	0.048	580	2400	2000

<sup>a</sup> Units are in mg/L.

<sup>b</sup> Pre-flood results are from the November 2009 sampling event.

<sup>c</sup> 2010 flood results from the June 2010 sampling event.

<sup>d</sup> 2012 results are from the December 2012 sampling event.



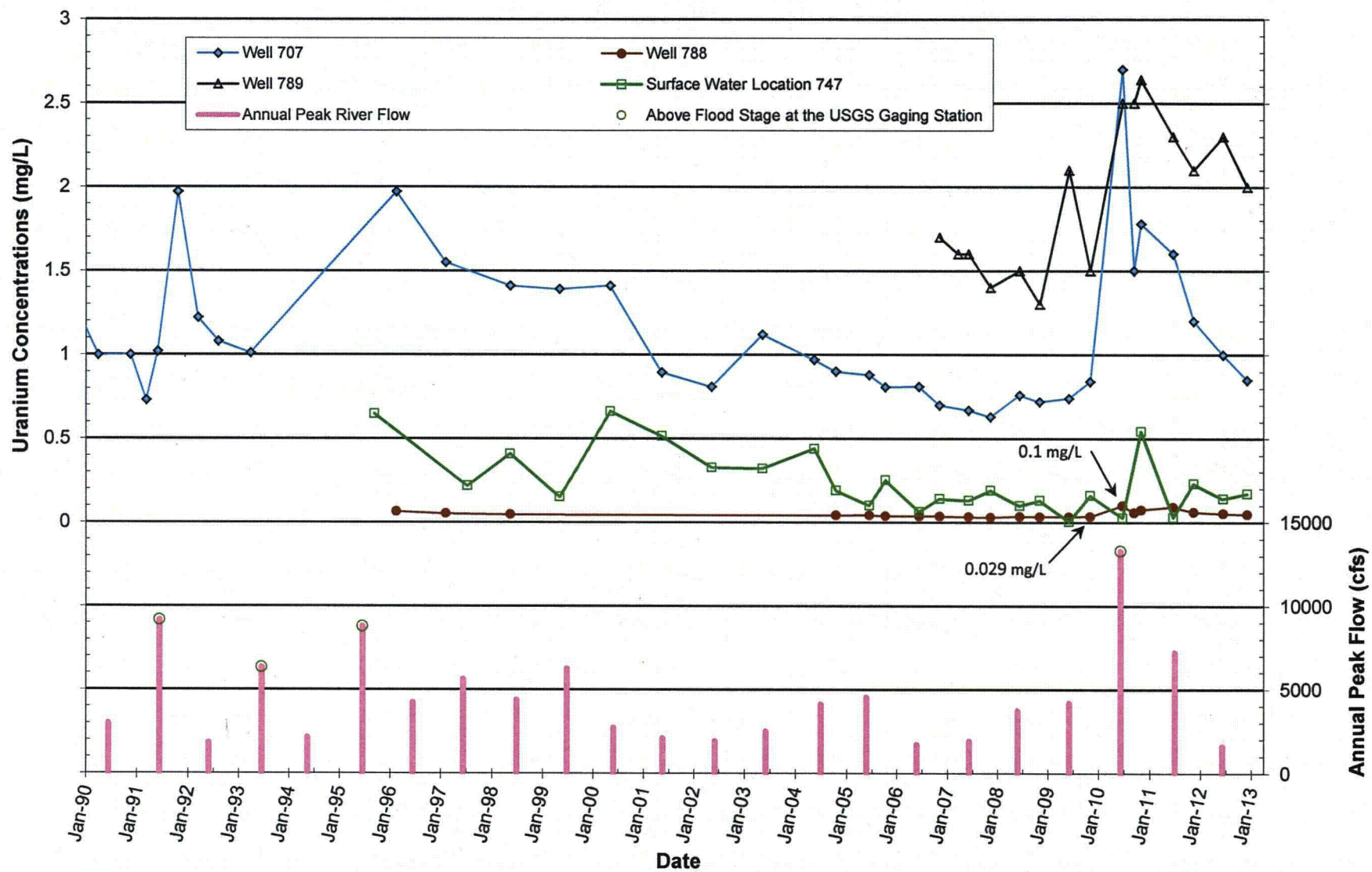


Figure 48. Uranium Concentrations and Maximum Little Wind River Stage



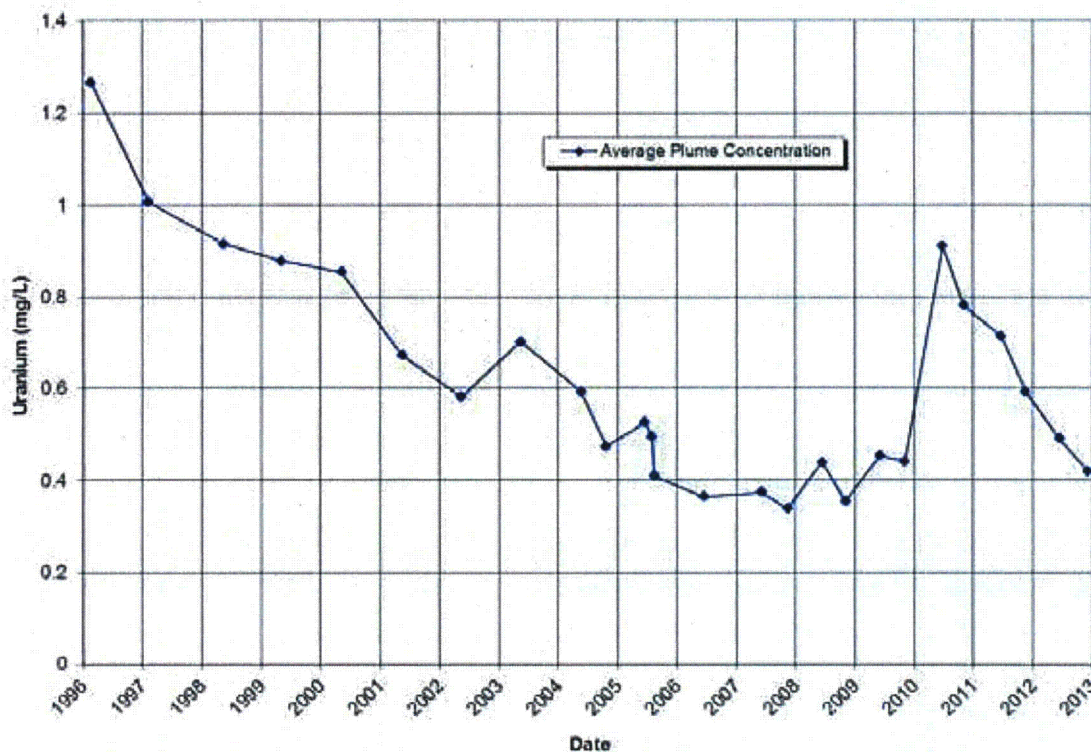


Figure 49. Average Uranium Concentration in Plume Wells

Overall, natural flushing (contaminant movement and removal via groundwater flow) in the surficial aquifer is occurring; however, the rate of flushing does not currently appear to be fast enough to restore the aquifer within the 100-year regulatory time requirement. Several lines of evidence indicate that the natural flushing compliance strategy may not meet the 2089 target date. These include:

- Current plume configurations and magnitude developed from the 2012 enhanced characterization.
  - Uranium concentrations of 1.1 mg/L still exist on the former mill site, which indicates contaminant plume movement is retarded by aquifer properties and/or influenced by additional source.
  - Uranium concentrations in the center of the plume adjacent to the Little Wind River are greater than 2 mg/L, which is very high compared to the uranium standard of 0.044 mg/L.
- Recently completed groundwater modeling indicates aquifer restoration will take longer than 100 years from the present.
- Other UMTRCA former uranium mill-sites with similar geology and contaminants are not cleaning up as predicted by groundwater modeling done to support a natural flushing compliance strategy.



- Time versus concentration graphs for average concentrations and for individual wells show that concentrations of contaminants are not declining as rapidly as in the past and/or have leveled out.
- Future flooding of the Little Wind River will likely cause an increase in contaminant concentrations in groundwater, even if the increase is relatively short-lived.
- Additional contaminants in the saturated and/or unsaturated zone may be acting as additional contaminant sources for elevated concentrations in groundwater.

Although the completion of natural flushing within the 100-year regulatory time frame is uncertain, additional information will be required to make a definitive decision on the natural flushing compliance strategy. A better understanding of the Riverton site, including aquifer properties, geochemistry, and potential additional contaminant sources, will be needed to support the natural flushing compliance strategy or to select a new compliance strategy. Some of this potential future work and information are identified in the Summary and Recommendations Section.



## 5.0 Summary and Recommendations

Verification monitoring results from 2012 verify that mill-related groundwater contamination continues to impact the surficial aquifer and oxbow lake, but institutional controls are in place and functioning as intended to protect human health and the environment from the groundwater contamination. In addition, verification monitoring results continue to verify that mill-related contamination has not impacted any potable domestic wells within the IC boundary, the semiconfined aquifer, the confined aquifer, the Little Wind River, gravel pit ponds, or the AWSS. Results from the AWSS flushing program provide evidence that the flushing program is effective in controlling the buildup of natural occurring radionuclides found in the source wells for the system.

The enhanced characterization conducted in 2012 resulted in a better understanding of uranium concentrations in the unsaturated zone soils and groundwater contaminant distributions. Uranium is present in higher concentrations in the unsaturated zone soils overlying the uranium plume than in the unsaturated zone soils overlying areas outside the contaminant plume. Although higher in the plume area, the range of labile uranium concentrations measured in the unsaturated soil are comparable to abundances of uranium in sedimentary rocks that make up the crust of the earth and may not be high enough to cause the increases that were observed in groundwater contaminant concentrations after the 2010 flood of the Little Wind River. Enhanced definition of groundwater plumes was obtained from the enhanced characterization effort, which also provided (1) evidence of the influence of the sulfuric acid plant discharge on the sulfate plume and (2) higher-than-expected uranium concentrations in the surficial aquifer on the south edge of the former tailings pile and in an area southwest of the primary uranium plume.

Although still above their respective MCLs, molybdenum and uranium concentrations in the surficial aquifer groundwater have returned to their pre-flood levels after spiking following the 2010 flood of the Little Wind River. However, numerous lines of evidence, including updated groundwater modeling, indicate that the rate of natural flushing is not rapid enough to meet the 100-year regulatory limit.

Although DOE obtained a better understanding of the site conceptual model, contaminant distributions, and properties of the unsaturated zone of the surficial aquifer at the Riverton site in 2012, additional work is needed to further define the conceptual model, to better understand geochemical processes that control contaminant fate and transport, to identify additional sources of uranium that are liberated during flood events, and to understand why uranium concentrations decline relatively quickly after flood events. This additional information will assist in making decisions for a path-forward compliance strategy. Recommendations for potential future work are listed in Table 16. DOE will prioritize the potential future work, will add medium- and high-effort work to future budgets, and will schedule the work. Low-effort work will be conducted under the current budget.



Table 16. Recommendations for Potential Future Work

Work Scope	Effort <sup>a</sup>	Purpose	Comments
Field Investigation			
Field observation of seeps.	Low	Assess groundwater discharge to the Little Wind River in accordance with the current site conceptual model.	Conduct during routine sampling.
Vertical measurements in wells and assessment of screened interval in the monitoring network.	Low	Determine if vertical contamination stratification exists and, if so, what are the impacts to the current understanding of the surficial aquifer contamination.	Vertical measurements of specific conductance can be conducted during routine sampling.
Additional water-level data loggers.	Low	Estimate irrigation infiltration impacts on groundwater flow and contaminant migration.	Wells 0826 and 0722R.
In situ measurements of groundwater flow direction.	Medium	Estimate flow direction in the surficial aquifer based on in situ measurements, and compare that with the flow direction based on water levels.	Research and purchase of equipment needed.
Install stilling well on the Little Wind River adjacent to well 0789.	Medium	Assess groundwater discharge to the Little Wind River in accordance with the current site conceptual model.	
Install stilling well on the Wind River with an adjacent monitoring well.	Medium	Evaluate interaction between the Wind River and the surficial aquifer.	
Additional field characterization with the Geoprobe; additional groundwater sampling around hot spots and the Little Wind River.	High	Better define the extent of groundwater contamination.	Work Plan required to define specific activities, objectives, and scope.
Additional field characterization with a drill rig, including soil/alluvial aquifer sampling below 5 ft, sampling of the clay/shale layer at the base of the alluvium. Perform lab experiments on samples.	High	Estimate the location of the sources of uranium and molybdenum that are mobilized during flood events. Determine the distribution of contaminants in saturated and unsaturated zone sediments.	Work Plan required to define specific activities, objectives, and scope.
Pilot tests or feasibility studies based on potential compliance strategy.	High	Determine feasibility of potential compliance strategies (if active remediation).	Work Plan required to define specific activities, objectives, and scope.
Laboratory Analyses			
Additional groundwater analyses: major cations/anions, total organic carbon, sulfide, and chloride.	Low	Better understand geochemical properties of the aquifers.	Can be conducted during routine sampling; analytical costs only.
Additional lab experiments on fine and coarse sediments in unsaturated zone.	Medium	Estimate source distribution in the unsaturated zone.	Perform tests on soil samples from 2012.
Assessment of sulfate in the semiconfined aquifer, including sulfur isotope analysis, additional chemical analyses.	Medium	Evaluate whether sulfate in the semiconfined aquifer is mill related.	Could be a High effort, depending upon the number and types of analyses.



Table 16 (continued). Recommendations for Potential Future Work

Work Scope	Effort <sup>a</sup>	Purpose	Comments
X-ray diffraction tests.	Medium	Identify clay and mineral types to assist in geochemical modeling, identifying contaminant sources, and assessing contaminant mobility.	Perform tests on soil samples from 2012.
Backscatter electron imaging and spectroscopy for mineralization analysis.	Medium	Identify mineral types to assist in geochemical modeling, identifying contaminant sources, and assessing contaminant mobility.	Perform tests on soil samples from 2012.
Data Evaluation			
Flood frequency analysis of Wind River.	Low	Predict future flooding of the Wind River.	
Assessment of chloride concentrations in groundwater as a conservative tracer.	Low	Estimate irrigation infiltration impact on groundwater flow and contaminant migration.	
White Paper detailing compliance strategy options.	Medium	Present feasibility, requirements, and data gaps for each compliance strategy option under UMRCA to enhance communications with NRC and stakeholders.	Budgeted in 2013.
Geochemical and reaction path modeling.	High	Better understand geochemical processes that control fate and transport of site contaminants.	Work Plan required to define specific activities, objectives, and scope.
Additional groundwater modeling, coupled with geochemical modeling.	High	Better understand groundwater contaminant transport that includes groundwater flow and geochemical aspects.	Work Plan required to define specific activities, objectives, and scope.

<sup>a</sup> Low = less than 40 hours of labor or less than \$1,000 of cost. Medium = between 40 and 160 hours of labor or less than \$10,000 of cost. High = greater than 160 hours of labor or greater than \$10,000 cost.



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