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Pacific Northwest National Laboratory
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RCRA Groundwater Quality Assessment Report for Waste Management Area S-SX
(November 1997 through April 2000)

V. G. Johnson
C. J. Chou

July 2001

Prepared for the U.S. Department of Energy
under Contract DE-AC06-76RL01830

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Pacific Northwest National Laboratory
Richland, Washington 99352

Preface

This report was written to comply with the requirements stipulated in the Resources Conservation and Recovery Act (40 CFR 265, Subpart F) and in Washington State dangerous waste regulations (WAC 173-303). These regulations require groundwater monitoring at facilities that treat, store, transfer, and/or dispose of dangerous waste.

The regulated unit addressed in this report is one of seven single-shell tank waste management areas (WMA) at the Hanford Site located in south-central Washington State. The single-shell tanks contain radioactive high-salt defense waste generated during the chemical separation of weapons grade plutonium. Over half of the 149 individual single-shell tanks are known or suspected to have leaked. Spills associated with waste transfers within the WMAs have also occurred. Retrieval, processing, and final disposal and/or stabilization in place of these wastes will take place over the next 30 to 40 years.

Mobile tank waste constituents (e.g., technetium-99, hexavalent chromium, and nitrate) have appeared in some downgradient wells at five of the seven single-shell tank WMAs. Characterization of the groundwater and vadose zone are underway to evaluate the nature and extent of the subsurface contamination. The groundwater studies at the single-shell tank WMAs are part of the Hanford Groundwater Monitoring Project conducted by Pacific Northwest National Laboratory for the U.S. Department of Energy. Additional background information and related subsurface conditions at Hanford Site can be found in the annual groundwater report.

Summary

This report updates a continuing groundwater quality assessment for single-shell tank Waste Management Area (WMA) S-SX in the 200 West Area at the Hanford Site. The assessment was initiated in 1996 to evaluate the rate, extent, and concentrations of contaminants attributable to this Resource Conservation and Recovery Act (RCRA) regulated unit. During the report period (November 1997 through April 2000), four new groundwater monitoring wells were installed to evaluate vertical and areal extent of the contamination, hydrologic and tracer tests were completed to obtain better estimates of flow rate and direction, and sampling and analysis were performed to define contaminant concentrations. Major new findings include the following items:

- Groundwater contamination attributable to tank leaks or spills continues to persist in both the northern half of the WMA (S tank farm) as well as in the southern half (SX tank farm). The highest contaminant concentration (technetium-99 of 63,700 pCi/L versus the drinking water standard of 900 pCi/L) was found near tank SX-115 in the southwest corner of the SX tank farm. Based on the consistency in contaminant ratios and the direction and rate of groundwater flow, a source in the vicinity of tank SX-115 is the most likely source of the groundwater contaminant plume that occurs at the south end of WMA S-SX.

- Evaluation of changes in water-table elevations indicates a gradual shift in the direction of groundwater flow from the southeast to a more easterly direction. Hydrologic testing and tracer experiments suggest a contaminant plume originating in WMA S-SX should travel very slowly (30 to 50 meters per year). Monitoring results suggest the plume emanating from the south end of the WMA is relatively small or localized. New wells drilled in fiscal year 2000 and 2001 should help define the lateral or downgradient extent.

- Discrete depth sampling suggests mobile tank waste contaminants (nitrate, technetium-99, and tritium) are at the very top (upper 5 meters) of the aquifer in downgradient wells along the southeast side of the SX tank farm.

The groundwater quality assessment will continue with installation of additional monitoring wells to fill gaps in the network coverage, associated hydrologic testing, and additional discrete depth sampling. The groundwater assessment results will be integrated (by reference or inclusion) with RCRA Facility Investigation/Corrective Measures Study (RFI/CMS) in the report of findings of the vadose zone work conducted by CH2M HILL Hanford Group, Inc. for the U.S. Department of Energy's Office of River Protection.

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1.0 Introduction

This report presents the findings of a continued groundwater quality impact assessment at Waste Management Area (WMA) S-SX in the 200 West Area of the Hanford Site (Figure 1.1).

1.1 Background

Waste Management Area S-SX was placed into groundwater quality assessment monitoring status in June 1996. An initial assessment report, based on the results of a first determination, was issued in February 1998 and concluded the WMA was contributing to groundwater contamination (Johnson and Chou 1998). Thus, a continued assessment of the rate, extent, and concentration profiles of the contamination is required [see 40 CFR 265.93(7)]. Accordingly, an assessment plan (Johnson and Chou 1999a) was prepared to obtain the data needed to determine the rate and extent of contaminant migration and their concentrations in the groundwater. The groundwater assessment for WMA S-SX is being conducted concurrently and in coordination with the vadose zone investigations for the Resource Conservation and Recovery Act (RCRA) Facility/ Corrective Measures Study (RFI/CMS), as described in Tri-Party Agreement Milestone M-45 (Ecology et al. 1998). The RFI/CMS work is being conducted by CH2M HILL Hanford Group, Inc. (Tank Farm Vadose Zone Project) for the Office of River Protection, Department of Energy, in response to Tri-Party Agreement Milestone M-45. Summary information on assessment results is included in quarterly reports to the Washington State Department of Ecology (Ecology) and annually, as required, in the groundwater monitoring annual reports, e.g., Hartman et al. 2000.

1.2 Scope

Only new water quality data and hydrologic testing results obtained subsequent to the first assessment are included in this report. Hydrogeology of the site, stratigraphy, waste site descriptions, and contaminant hydrology were described in the first assessment report (Johnson and Chou 1998) and addendum (Johnson and Chou 1999b) and in the updated assessment plan (Johnson and Chou 1999a). Therefore, the scope of this report is limited to evaluation and interpretation of new data acquired from

1. four new wells installed during October 1999 through February 2000
2. resampling of three older tank farm wells
3. special analyses conducted on selected wells
4. additional quarterly sampling from the existing network since November 1997 through April 2000.

Supporting information (e.g., drilling information, hydrologic testing raw data, computer simulation runs) for this report are available in the project files of the Hanford Groundwater Monitoring Project at Pacific Northwest National Laboratory (PNNL) and in the borehole data packages for the new wells that were drilled during the report period (Horton and Johnson 2000). Integration of tank farm vadose zone study results and groundwater assessment findings will be addressed in the initial RFI/CMS report.

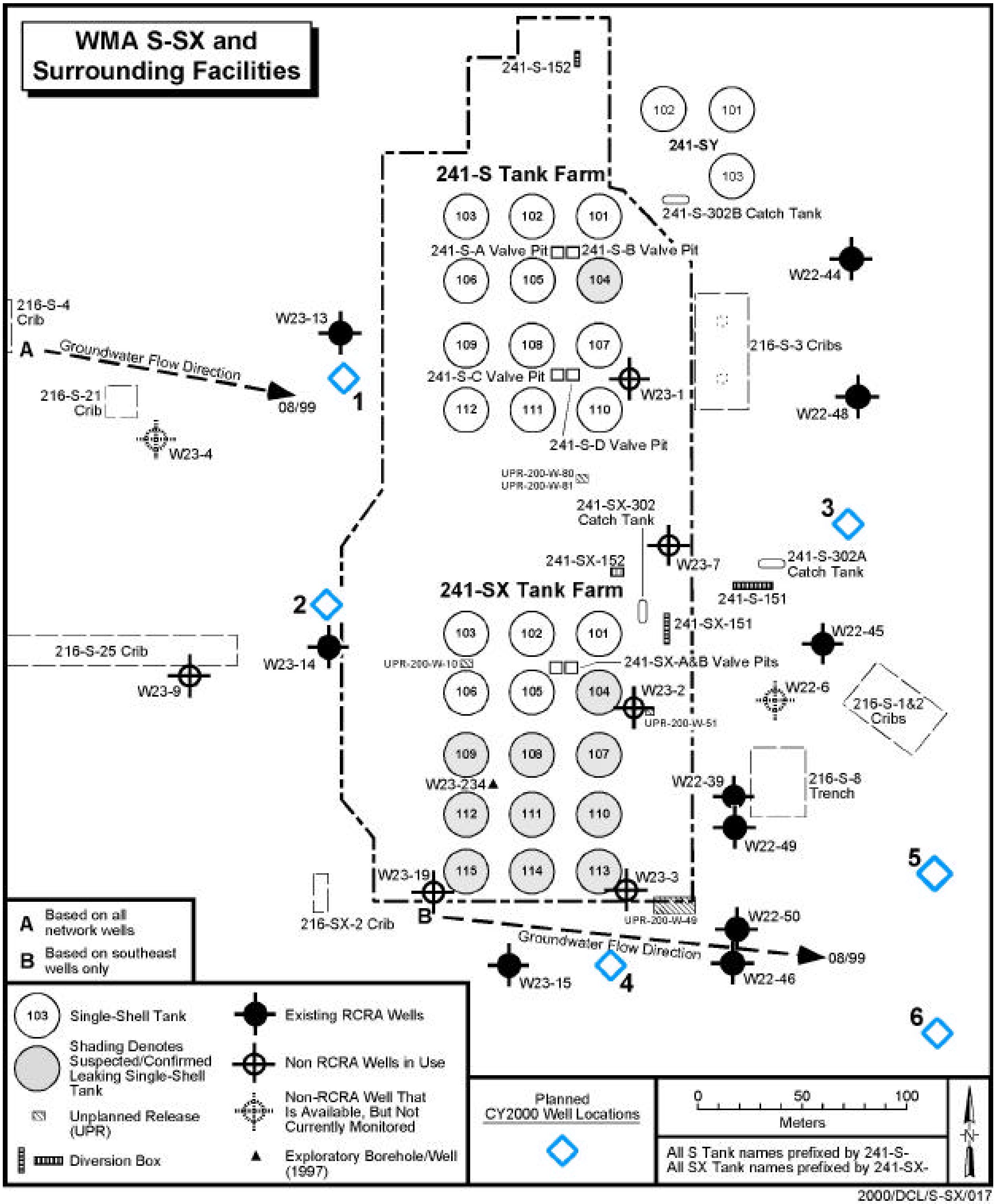

Figure 1.1. Waste Management Area S-SX, Surrounding Facilities, and Well Locations

1.3 Report Organization

Organization of this report is based on the objectives for the continuing assessment, which are to determine the rate and extent of migration and concentration of groundwater contamination. Accordingly, Chapter 2.0 addresses the rate of groundwater movement and direction of flow based on hydrologic data acquired during the report period. Chapter 3.0 addresses areal and vertical extent of contamination, contaminant concentration, and contaminant types based on new observations made during drilling of new characterization and monitoring wells for this assessment. Chapter 4.0 provides an assessment of the theoretical efficiency of the monitoring well network to detect contaminants originating from within the WMA. Chapter 5.0 provides information on the highest contaminant concentrations found at the WMA. Chapter 6.0 presents conclusions regarding the rate and extent of contaminant migration, possible source areas, and the likelihood of detecting groundwater contamination that could arise from this WMA in the future.

2.0 Rate and Direction of Groundwater Flow

The rate of groundwater movement beneath and in the vicinity of WMA S-SX is estimated from classical methods (Darcy equation), borehole tracer dilution tests, and observation of contaminant plume movement and tracer drift test arrival times.

2.1 Darcy Velocity

The Darcy equation for estimating velocity (v) requires measurement of hydraulic conductivity (K), effective porosity (n_e) and hydraulic gradient (i). The velocity is calculated from the following relationship:

$$v = Ki/n_e$$

For the WMA S-SX assessment, new hydraulic conductivity data were obtained from slug tests and drawdown tests conducted in the new wells installed for this study and in selected existing wells. Effective porosity was determined using tracer drift and pumpback test methods as described in PNNL Procedures for Groundwater Investigations (PNL-MA-567, AT-7). Hydraulic properties determined for this study are presented in Appendix A.

Variation in hydraulic conductivities among the existing and new wells tested (see Table 2, Appendix A) indicates that the aquifer is not homogeneous in the study area. The results shown in Appendix A also suggest the average groundwater velocity should be very low in the study area, as previously reported (Connelly et al. 1992). This is consistent with estimates (25 to 50 meters per year) based on assumed arrival of tritium in downgradient wells from upgradient sources (Johnson and Chou 1999).

2.2 Borehole Dilution Testing

Borehole dilution and pumpback tests were conducted in the three new RCRA wells. These tests permitted some inferences about flow rate as well as aquifer homogeneity and effective porosity. After introduction of the bromide tracer into the borehole (PNL-MA-567, AT-7), continuous measurement of the bromide concentrations were made using a downhole bromide sensor and data logger. Five probes positioned about 1 meter apart were used to cover the 4.6 meter screened interval. This test allowed direct observation of the effect of lateral groundwater flow through the screened interval of the borehole. It also provided an indication of the uniformity of flow over the screened interval. Details of the test, computations, and raw data are included in the Groundwater Monitoring Project files (PNNL, Sigma V).

Apparent velocity is calculated from the relationship between tracer concentrations, C_o (at time zero) and C_t (at time t), borehole volume (V), volumetric flow through the screen (Q) and time (t)

$$\ln (C_t/C_o) = (Q/V) * t$$

The velocity (v_w) in the well bore can be calculated from the above relationship by solving for Q and dividing by the longitudinal cross sectional area (A) of the well screen:

$$v_w = Q/A = [\ln (C_t/C_o) * V/t] /A$$

The apparent velocity in the aquifer, v_a (based on v_w adjusted for porosity and distortion effects), for the three wells tested and associated conditions are summarized in Appendix A, Tables 3 and 4. Results based on the above relationship vary but support the low Darcy velocities and inferences based on contaminant plume arrival times between monitoring wells.

One important additional finding from the bromide tracer tests was that in well 299-W22-48, only the upper 1.5 meters of the screened interval (4.5 meters) exhibited significant dilution of the tracer (i.e., flow of water through the screen) during the period of observation. The other two wells (299-W22-49 and 299-W22-50) located 175 meters and 225 meters to the south, respectively, exhibited a relatively uniform movement of water over the screened interval (all three wells were completed with 4.5 meters of submerged screen). The non-uniformity with depth noted in well 299-W22-48 may be due to the non-homogeneous nature of the Ringold Formation in this area. Such hydraulic irregularities have been previously reported for the Ringold at the north end of 200 West Area (Swanson 1994, pages 81 and 82; Lindsey and Mercer 1994, page 54).

A separate topical report on the tracer and hydraulic testing conducted for the 200 West Area single-shell tank WMA assessment sites will be issued in 2001.

2.3 Large Scale Bromide Tracer Drift Test

A volume of 16,000 liters of a 60-ppm bromide solution (in Columbia River water) was injected into the top of the aquifer beneath the SX tank farm in March 1999, just prior to abandonment of borehole 41-09-39 (now named well 299-W23-234, Figure 1.1). The tracer was injected into a shallow (1.5-meter) screened interval in an attempt to simulate a large area source that had just entered the aquifer. The total dissolved solids content of the bromide solution matched the ambient groundwater. An initial tracer patch of 20 meters in diameter, or about the diameter of a single-shell tank, was intended. The primary objective was to test the efficiency of the downgradient monitoring wells to detect a simulated leak from a tank source. The elapsed time between when the tracer was injected and when it first appears in a downgradient monitoring wells should also indicate flow rate in that area of the WMA. Bromide measurements in downgradient wells are made by special request on samples collected during the routine RCRA quarterly sampling of the monitoring network wells. Over a year and a half has passed since injection of the tracer. No evidence of the tracer arrival has yet appeared in any of the wells downgradient from the point of injection (i.e., wells 299-W23-19, 299-W23-15, 299-W22-50, 299-W22-49, 299-W22-46, 299-W22-39, and 299-W22-45).

The absence of bromide from the tracer drift test in any downgradient well is consistent with the computed Darcy velocities and the contaminant plume arrival times. For example, the nearest distance to a downgradient well from the point of injection is ~100 meters. Thus at 30 meters per year, it should take over 3 years before the tracer arrives at this well. At 50 meters per year, it should take 2 years or another

0.5 years before any tracer appears in downgradient wells. The absence of the tracer tends to support the slow travel times indicated above, or indicates that the tracer plume has not migrated in the direction of a monitoring well.

2.4 Flow Direction

The direction of groundwater flow was estimated based on the gradient in the water-table elevations in the S-SX network monitoring wells. This approach assumes the aquifer is homogeneous. Because there is evidence that the aquifer is non-homogeneous, this limitation must be kept in mind when applying the gradient analysis approach to estimate flow direction. A general flow direction may be estimated over the study area, but at any specific location, perturbations may occur in the local flow direction due to localized low permeability zones.

Recognizing the above uncertainty, trend surface analysis was applied to the water-table elevation gradient for various combinations of wells (see Appendix B). Annual water-table elevation measurements from 1992 to the present were selected for the same month of the year (August) to minimize atmospheric disturbance effects (barometric pressure changes can cause fluctuations in the static water level in a well and these effects are at a minimum during the late summer). Three combinations were evaluated: (1) the S tank farm and vicinity; (2) SX tank farm; and (3) the WMA as a whole. Results are presented in Appendix B, Tables 3, 4, and 5.

A change in the direction of groundwater flow from southeast to a more easterly direction over time is evident in all three cases evaluated. Most of the shift in flow direction occurred in the northern part of the WMA. This is because until June 1995, wastewater was discharged to a ditch along the northwestern edge of the WMA. The ditch caused a localized groundwater mound at that location, causing a southeasterly flow direction beneath the WMA. Thus, prior to 1995 the prevailing direction of groundwater flow was more southeasterly. At the present time, the trend surface results for all the network wells combined suggests groundwater is flowing in nearly a due east direction.

The larger scale water-table map of WMA S-SX (Figure 2.1) and the surrounding area suggests a flow direction that is a little more east-southeast than indicated by the localized trend surface analysis for WMA S-SX. The apparent tritium plume superimposed on the water-table map also seems consistent with the flow direction suggested by the larger scale water-table map. The difference in the trend surface results based on a relatively close grouping of network wells (WMA S-SX network) versus the larger area water-table map may be one manifestation of the effect of a non-homogeneous aquifer.

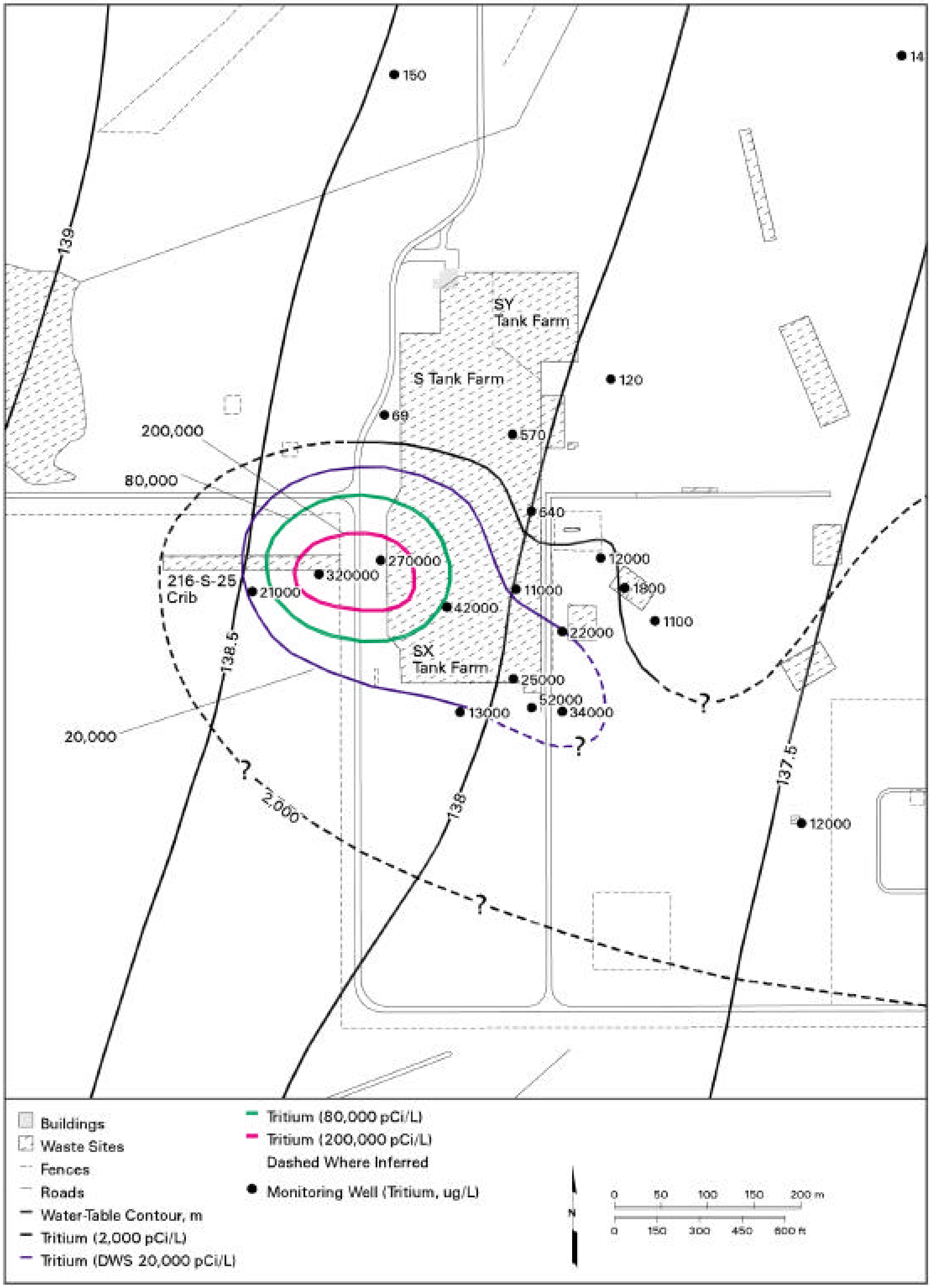

Figure 2.1. Tritium Plume and Water-Table Elevation Map for WMA S-SX and Vicinity (1999)

3.0 Extent of Contamination

Evaluation of the extent of contamination involves investigation of the depth distribution of contaminants, the areal distribution pattern, and the type and concentration of contaminants in the groundwater. Results of depth sampling during installation of new RCRA groundwater monitoring wells, and the completion of vadose zone exploration boreholes as groundwater monitoring wells, provided new insights into the occurrence and nature of groundwater contamination attributable to WMA S-SX.

3.1 Theoretical Considerations

Contaminants entering the surface of a homogeneous, unconfined aquifer can be dispersed downward as well as laterally and longitudinally. The degree of vertical spreading varies depending on the dispersivities and the hydraulic gradient or driving force (local recharge or net drainage to the aquifer from precipitation events) and whether or not the density of the waste fluid is greater than the ambient groundwater.

Very few simulations of vadose zone/groundwater coupled solute transport have been performed for 200 West Area waste sites. However, coupled modeling was conducted in an attempt to simulate movement of wastewater and contaminants through the vadose zone beneath the 216-U-17 crib in the 200 West Area (Reidel et al. 1993). These simulations were run with the computer code PORFLO3 and used local aquifer conditions with up to 10 centimeters per year of recharge through a gravel-covered surface (similar to a tank farm surface). The liquid loading per unit area was similar to a large tank leak. Results indicated that after passing through the vadose zone (70 meters thick), a technetium-99 and uranium concentration maximum developed from about 10 to 20 meters below the water table 40 to 50 years after discharge to the crib ceased. At a relatively short lateral distance away (50 to 100 meters), the groundwater plume was pushed below the water table (no contaminant from 0 to 10 meters below the water table). The simulation also indicated that once the source passed through the vadose zone it "bled" slowly into the aquifer for tens of years. Slow drainage from the soil column to groundwater was also one outcome of the vadose transport modeling of a tank leak in the SX tank farm (Ward et al. 1997).

Departure from the theoretical depth distribution in a homogenous aquifer may occur depending on the nature of the aquifer host rock. As noted in Chapter 2.0, there are indications of inhomogeneities in the Ringold Formation in the study area. For example, if a relatively impermeable zone lies just beneath a more permeable upper zone at the water table, movement with depth would be restricted. In this case, a shallow contaminant zone at the surface could result. The opposite could also occur with a less permeable zone at the water table and a more permeable zone beneath it. In that case, the local variation in lithology would result in a subsurface maximum in the contaminant profile. These hypothetical variations cannot be modeled so there is no alternative to direct observation (i.e., discrete depth sampling from drilled boreholes and monitoring wells within a known plume area). Such features could also result in deviations in predicted (horizontal) plume flow directions due to lateral preferential flow paths. Contaminant dispersion models of lateral movement are also of limited value in such cases. Field observations made during the reporting period may reduce these uncertainties at WMA S-SX, discussed as follows.

3.2 Vertical Distribution Data

Sampling with depth was conducted at four new wells during the report period. One new ground-water well (299-W23-19) was initially drilled as a vadose zone characterization borehole adjacent to tank SX-115. This well apparently intersected a vadose zone plume that enters groundwater near the well. Both a shallow temporary and a deeper permanent completion depth provided contaminant depth information in groundwater beneath this known vadose zone source location. Another well was drilled to the bottom of the aquifer, sampled at seven discrete depths, and was then completed as a shallow monitoring well (299-W22-50) as described in the assessment plan (Johnson and Chou 1999a) and borehole data package report (Horton and Johnson 2000). In addition, samples of near-surface water and samples from the maximum depth drilled (~7 meters) were collected during drilling of two other new WMA S-SX wells. Results from these sampling efforts provide new depth distribution information. The well completed inside the SX tank farm and the new downgradient RCRA wells are discussed separately in the following sections.

3.2.1 Well 299-W23-19 (Tank SX-115)

This well was initially completed with a temporary 1.5 meter screen. Both a passive surface sample (KABIS[1] sampler) and a pumped sample (1 meter below static water level) were obtained before the well was deepened and completed with a permanent 9.1 meter screen. Pumped samples were obtained from the longer screened interval with the pump set at 2 meters below static water level. Concentrations of key contaminants and major cations and anions are shown in Table 3.1. Concentrations do not seem to vary much under very different well screen lengths and sampling depths.

The technetium-99 concentration shown in Table 3.1 is the highest yet observed on the Hanford Site. The relatively high chromium and nitrate are consistent with the tank waste signature expected as discussed in previous reports (Johnson and Chou 1998; Johnson and Chou 1999a,b). Other major constituents of concern for tank waste (americium-241, cesium-137, iodine-129, neptunium-237, plutonium-239/240, strontium-90) were all non-detects in the samples collected in October 1999. Tritium was moderately elevated, but it is not clear if this is from upgradient sources or a combination of tank waste and upgradient sources. A relatively low ratio of tritium/technetium-99 (~1) is consistent with a tank waste source as compared to a residual upgradient evaporator source (e.g., the 216-S-25 crib) as discussed in Johnson and Chou (1998).

The sodium relative to calcium (see Table 3.1) is also noteworthy. The much higher concentration of calcium relative to sodium is indicative of a small volume, high salt (sodium nitrate matrix) release event. Thus, even though sodium is the dominant cation in tank waste, as it migrates through the vadose zone it can exchange with the calcium in the soil so that the pore fluid that eventually migrates to the water table is enriched in calcium, depicted as follows:

$$2\ Na^{+}(\text{pore fluid}) + Ca\text{-Soil} => Ca^{++}(\text{pore fluid}) + 2\ Na\text{-Soil}$$

This is another piece of indirect evidence consistent with a tank leak or spill source.

[1] KABIS is a registered trademark of SIBAK Industries Limited, Peoria, Illinois.

Table 3.1. Observed Contaminant Concentrations in Shallow and Extended Well Completions Near Single-Shell Tank SX-115 (Well 299-W23-19)

Constituent (unit)	Shallow Temporary Screen (1.5 m)		Permanent 9.1 m Screen	
	October 1999[a]		March 2000	June 2000
	Passive (Kabis, 0–6 cm)	Pumped[b] (1 m)	Pumped[c] (2 m)	Pumped[c] (2 m)
Technetium-99 (pCi/L)	48,050	39,000	52,300	63,700
Chromium (μg/L)	84	63	90	87
Nitrate, as NO_3^- (mg/L)	560	434	491	562
Tritium (pCi/L)	92,000	91,000	95,800	92,000
Specific Conductance (μS/cm)	1,199	1,003	968	1,237
Sulfate (mg/L)	18	16	18	17
Calcium (mg/L)	118	96	127	120
Magnesium (mg/L)	39	32	41	40
Sodium (mg/L)	34	34	42	43
Chloride (mg/L)	15	12	16	16

(a) Values reported represent duplicate averages.
(b) 1.5 m screened interval with pump intake set at 1 m below the static water level.
(c) 9.1 m screened interval with pump intake set at 2 m below the static water level.

In addition to the analyses discussed above, an effort was made to preconcentrate cesium-137 from a large aliquot of water collected during the initial sampling event at 299-W23-19. A lower detection limit for cesium-137 in groundwater was desired because previous observations based on spectral gamma logging of boreholes in the S and SX tank farms suggested that some cesium-137, a major tank waste constituent of concern, may have migrated through the vadose zone to groundwater. While cesium-137 is generally believed to be immobile, there was speculation that a more mobile species or complex may exist, or that some fraction of the cesium-137 may be associated with a colloidal phase. Even a small amount of cesium-137 in groundwater could thus be significant in understanding transport mechanisms. Thus, to determine if low concentrations of dissolved cesium-137 might be present, an 18-liter water sample was passed through a 0.4 μm filter and then through an Empore® cesium disk consisting of potassium ferrocyanide (KFC) embedded in a solid matrix. Both the pre-filter and the cesium disk were analyzed by gamma energy analysis (Savannah River Technology Center). This effort resulted in an apparent cesium-137 concentration of 0.01 ± 0.001 pCi/L. Such a low concentration is at a level where even a little surface contamination can affect the results. The significance of this observation is that even at a location directly beneath a known tank leak, where breakthrough of mobile contaminants to groundwater has been documented, there is little, if any, evidence of cesium-137 in groundwater. Technetium-99 was found in the soil column all the way down to near the water table at this location. While a colloidal fraction that escapes the filtration and KFC filter could be present, it seems highly unlikely. And even if such a fraction did exist, the cesium-137 concentration would have to be less than the detection limit (~2 pCi/L) for the total cesium-137 method used for the routine laboratory analyses.

That is, the water samples for direct GEA are unfiltered and are either analyzed directly in a Marinelli beaker or after evaporation of a 2-liter sample of unfiltered water down to 500 milliliters. Thus, even a colloidal fraction, if present, would be accounted for by the direct count method.

3.2.2 New Downgradient RCRA Monitoring Wells

Depth data were obtained during the drilling process at the three new RCRA wells installed during the report period. Wells 299-W22-48, 299-W22-49, and 299-W22-50 are discussed separately below. Details of the drilling and completion are described in Horton and Johnson (2000).

Well 299-W22-48. Slurry water was obtained by bailing after the drive casing penetrated the aquifer about 0.6 meter during the cable tool drilling process for this well. The highly turbid slurry water was allowed to settle first and then the supernate was drawn off and filtered for analysis. An attempt was made to sample from the bottom of the drilled depth of ~7 meters below the water table using a packer and pump assembly. Excessive inflow of sand, however, prevented placement of the pump. Thus only the 0.6-meter depth and the depth sampled at 2.3 meters below static water level (after the screen was installed) were obtained at this well. Results are shown in Table 3.2. There was no indication of higher concentrations of mobile tank waste constituents (chromium, nitrate, or technetium-99) at the surface as compared to results from deeper in the aquifer (based on the pumped sample from 2.3 meters in the screened interval). Concentrations of carbon tetrachloride, chromium, nitrate, technetium-99, and uranium were higher at the deeper, 2.3-meter sample depth as compared to the near-surface, 0.6-meter depth.

Well 299-W22-49. This new well was drilled in the same manner as described for well 299-W22-48. However, in this case, sample depths of both 0.5 meter and 6.7 meters (maximum drill depth) were obtained. The intermediate depth (2.3 meters) was obtained by pumping from within the 4.6 meters screened interval. Results are shown in Table 3.2. Only very low concentrations of tank waste indicators were detected and, as with well 299-W22-48, there is no indication of high concentrations of tank waste constituents at the surface of the aquifer. Tritium is present in this well and appears to be uniformly distributed with depth over the interval tested. The much higher tritium concentration relative to technetium-99 suggests a non-tank farm origin of the tritium.

Well 299-W22-50. This new well was drilled nearly to the bottom of the aquifer for vertical characterization purposes and was then completed as a standard shallow-depth monitoring well (see Horton and Johnson 2000 for drilling and completion details). Groundwater samples were collected at seven discrete depths beginning with the top ~0.2 meter. One of the discrete depths was just above the Ringold lower mud unit and one was below the lower mud unit. The Ringold lower mud unit formed a tight seal around the primary drive casing isolating the aquifer above the mud from that below. A head difference of approximately 1 meter (lower below the mud than above it) was observed, indicating good isolation was obtained prior to sampling the lowest test zone.

Table 3.2. Depth Distribution of Key Contaminants and Hydrochemical Parameters

Well	Sample Date	Depth (m)	Mode[(a)]	Contaminants					
				^{99}Tc (pCi/L)	NO_3 (μg/L)	Cr (μg/L)	^3H (pCi/L)	U (μg/L)	CCl_4 (μg/L)
299-W22-48	10/26/99	0.6	DT/B	39.5	17,132	3.2U	122U	0.2	0.4
	03/30/00	2.3	S	720	18,593	7.1	249U	3.23	4
299-W22-49	11/04/99	0.5	DT	32.5	13,546	3.2U	22,000	0.82	0.6
	03/30/00	2.3	S	58.3	9,296	4.6U	22,000	3.27	6
	11/08/99	6.7	DT	2.96U	7,880	3.2U	18,900	0.92	1
299-W22-50	11/23/99	0.2	DT/B	4,240	57,991	3.0U	31,400	0.78	13
	04/03/00	2.3	S	3,230	30,102	10.4	24,200	4.29	11
	11/29/99	6.7	DT	812	12,838	3.0U	19,900	3.34	5.6
	12/14/99	11.9	DT	7.03U	2,125	3.0U	969	1.09	0.94
	12/15/99	28.7	DT	0U	1,151	3.0U	304	0.58	1.5
	12/17/99	53.0	DT	0U	3,187	3.0U	185U	0.79	5.6
	12/22/99	67.7	DT	0.577U	12,838	3.0U	0U	0.43	0.89
	01/12/00	99.4	DT	0U	4,869	3.0U	0U	30.90	0.23

Well	Hydrochemical Parameters								
	pH	Conductivity (μS/cm)	Alkalinity (μg/L)	SO_4 (μg/L)	Cl (μg/L)	Na (μg/L)	Ca (μg/L)	Mg (μg/L)	Na/Ca
299-W22-48	7.97	263	74,000	21,300	6,910	26,300	19,400	5,070	1.36
	8.59	295	---	19,200	5,900	27,000	22,200	6,910	1.22
299-W22-49	8.94	245	90,000	13,900	5,330	25,600	16,200	5,070	1.58
	9.09	240	---	11,900	2,800	23,300	17,600	5,960	1.32
	8.1	244	86,000	15,400	3,660	26,000	16,400	5,320	1.59
299-W22-50	---	---	100,000	14,200	4,800	28,200	23,200	7,300	1.22
	8.14	278	---	13,400	3,100	23,900	22,600	7,100	1.06
	8.1	235	101,000	12,500	2,500	20,400	17,800	6,020	1.15
	8.2[(b)]	228	106,000	14,400	3,100	11,600	26,300	9,250	0.44
	7.9[(b)]	242	114,000	14,400	4,400	12,700	28,700	10,200	0.44
	7.9[(b)]	307	126,000	16,100	15,200	14,300	33,400	12,400	0.43
	7.7[(b)]	323	115,000	19,300	10,000	15,500	33,000	12,500	0.47
	8.5[(b)]	234	96,000	18,900	5,800	16,600	20,200	8,010	0.82

Note: U denotes analytical result is not detected.
(a) S = Sample collected by pumping from 4.5 m screened interval.
DT = Sampled during drilling using temporary pump/screen and packer assembly.
DT/B = Bailed during drilling.
(b) Laboratory result.

An air rotary drilling method was used to advance the borehole. When a desired depth was reached, the drill string was replaced with a submersible pump and packer assembly consisting of a 1.5 meter length of slotted PVC that served as a temporary screen. The inflatable packer was used to isolate standing water in the drive casing from the water pumped to the surface. Water was purged until stabilization of indicator parameters (pH, specific conductance, and temperature). Purge volumes were on the order of 400 liters. The water samples were filtered in the field to remove particulates. Due to the copious amounts of air introduced to the formation, it was not possible to obtain meaningful Eh and dissolved oxygen data. Also, samples of produced water were collected at 6.1-meter intervals during drilling operations. These samples were obtained just after a new section of pipe normally in 6.1-meter lengths was added. The short downtime for pipe addition allowed water to seep into the bottom of the open hole. When the air was turned back on, the accumulated water was blown out and sampled. (During the drilling process, the air pressure prevented water from entering the drill pipe and thus little, if any, free liquid was present in the cuttings). The water samples obtained from the drill pipe addition step were filtered and analyzed immediately for nitrate using a HACH® portable spectrophotometer. Selected samples were also analyzed for technetium-99 using a field screening method. Nitrate was the primary tank waste indicator used to track changes with depth as the drill bit was advanced. This provided the opportunity to stop and collect a pumped sample if unexpectedly high concentrations were encountered.

Numerical results from the pump and packer sample depths are shown in Table 3.1. Four prominent contaminants are plotted as a function of depth below static water level in Figure 3.1. The field nitrate data is shown in Figure 3.2 along with the results for the samples collected with the pump and packer assembly. The additional depth data provided by the air lifted samples analyzed in the field greatly enhances the depth profile based on the seven discrete depth pump and packer sets by filling in the gaps between the more widely spaced pump and packer sample depths. Overall, the drill and test sampling results meet the primary objective of determining whether or not contaminants are distributed deeply in the aquifer.

Figure 3.1 shows that nitrate, technetium-99, and tritium decline rapidly over the first 10 meters, with the highest concentrations at the top of the aquifer. There is low but detectable carbon tetrachloride down to the top of the mud unit located 75 to 80 meters below the water table. However, carbon tetrachloride was undetectable below the Ringold lower mud unit. The profiles suggest two different plumes were intercepted: (1) a shallow-depth plume consisting of tank waste constituents and (2) a deeper plume containing carbon tetrachloride and nitrate from an upgradient source other than WMA S-SX. The latter is probably related to past-practice wastewater discharges from the Plutonium Finishing Plan that were routed to U Pond (located ~400 meters west of WMA S-SX). The large groundwater mound from this pond probably drove contaminated pond water deep into the aquifer, which then traveled southeast beneath the WMA.

Also, the apparent but moderate subsurface maximum in carbon tetrachloride and nitrate (based on the single depth sample) is supported by the trend in the more frequent field nitrate measurements shown in Figure 3.2. The air lift/field indictor measurement data thus helps to substantiate the results based on the more limited number of pump and packer sets. In addition, the air-lifted water (produced after addition of each new section of drill pipe) requires no downtime expense for sampling, whereas the pump and

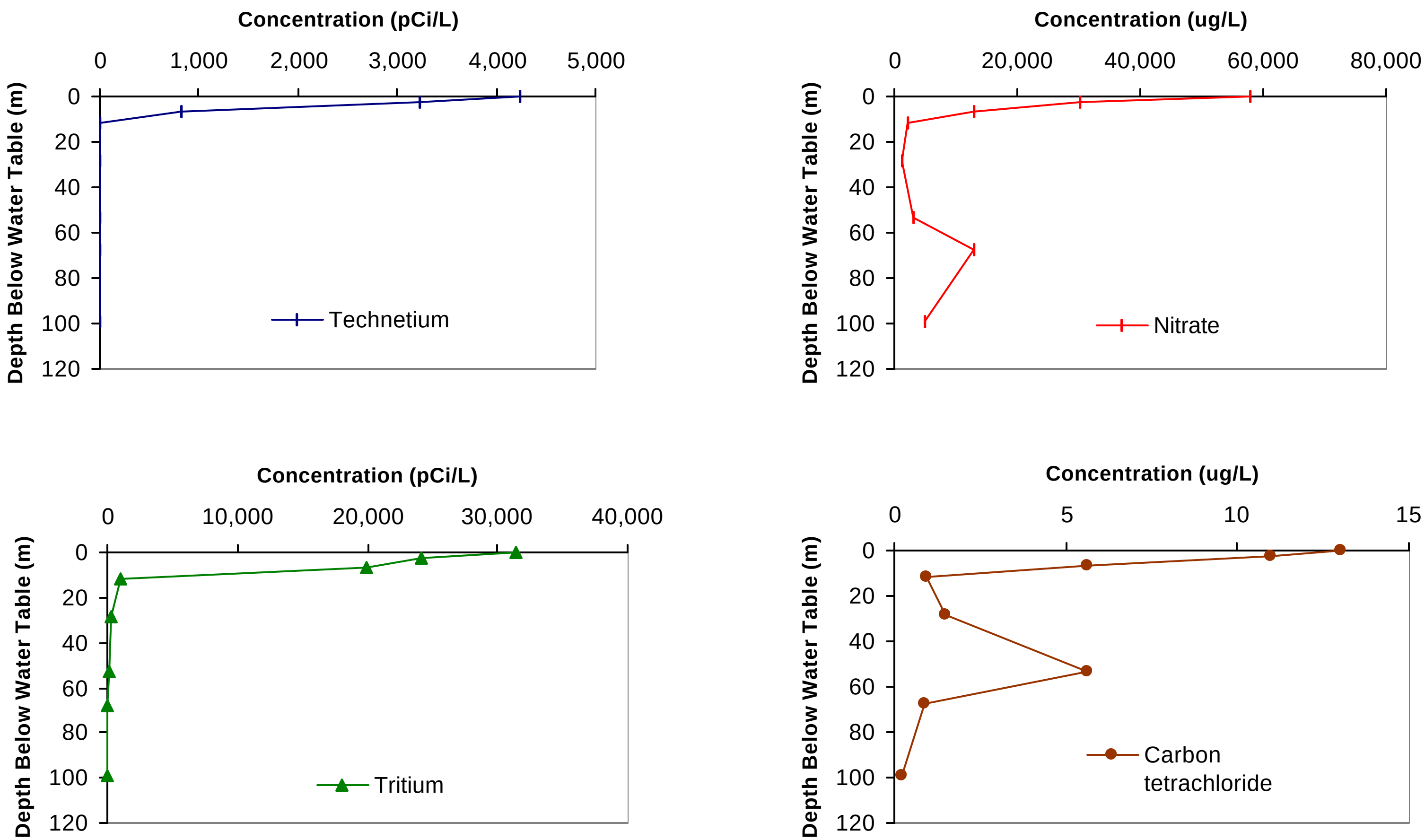

Figure 3.1. Groundwater Contaminant Concentration Depth Profiles for Well 299-W22-50

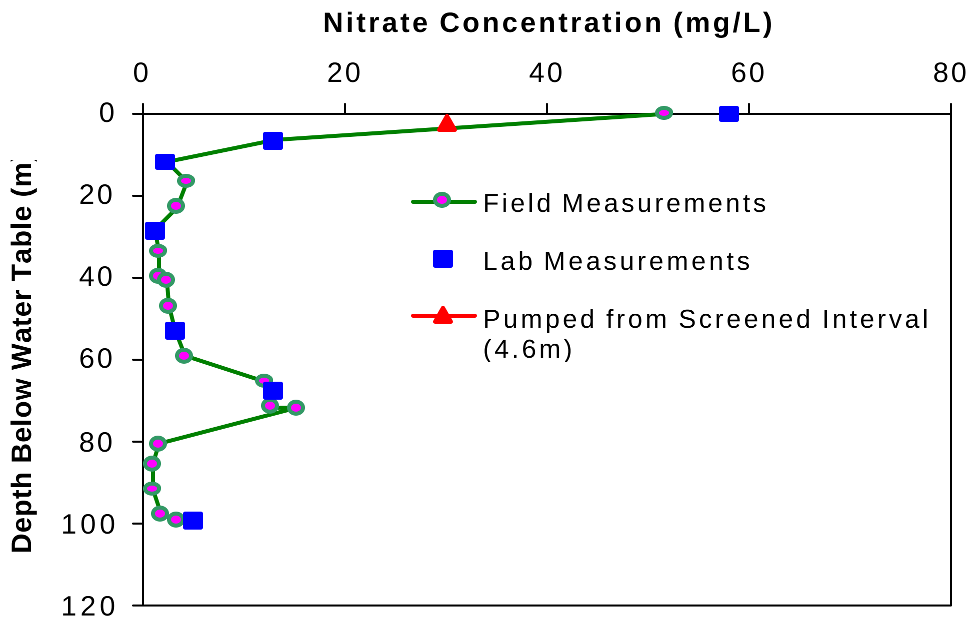

Figure 3.2. Nitrate Concentrations versus Depth at Well 299-W22-50 for both Air Lifted and Pumped Samples

packer sets are time consuming and require costly standby time for the drilling rig and crew. Combining the two sampling methods optimizes the depth information and provides a means to select depths for more detailed sampling with the pump and packer. Due to the large amounts of air introduced to the aquifer, however, dissolved oxygen and redox measurements cannot be made. Also, the carbon tetrachloride concentrations may be lower than actual due to the air stripping effects, and, therefore, they should be regarded as minimum concentrations.

The generally low electrical conductivity (EC), as shown in Table 3.2, is attributed primarily to the large volumes of low conductivity cooling water (Columbia River water with an EC of ~140 ì S/cm) that were discharged to nearby U Pond in the past. The average natural EC background for groundwater in
average groundwater and pond water would
yield a theoretical EC value of (0.5*140 + 0.5*360) = 250 ì S/cm. The conductivity values shown in Table 3.2 are just above or below the theoretical 50:50 mixture value, indicating that pond water (and any associated contaminants) invaded the entire depth of the unconfined aquifer at this location. The occurrence of carbon tetrachloride and nitrate at depth (see Figure 3.1) is consistent with the inference based on EC versus depth for well 299-W22-50 (see Table 3.2). Carbon tetrachloride and nitrate were probably dissolved in the large volumes of cooling water from the Plutonium Finishing Plant that were routed to U Pond until 1985.

One interesting observation is the moderately elevated uranium at the greatest depth. This occurrence is most likely related to a natural source of uranium since there are no other indicators of Hanford waste (e.g., tritium) that occur with it. Isotopic analysis of the uranium by inductively coupled plasma (ICP) mass spectroscopy could confirm the suspected natural origin by comparing the relative abundances of uranium-235, uranium-236 (produced in a nuclear reactor; does not occur naturally), and uranium-238.

3.3 Areal Distribution

It is difficult to display contaminant data as concentration contours because of the dynamic nature of leaks and disposal history in the vicinity of WMA S-SX. The maximum concentrations of the major mobile tank waste contaminants for the current report period, November 1997 to April 2000, are summarized in Table 5.1. Time series plots (Figure 3.3) are included for selected wells to illustrate concentration dynamics since 1992 (beginning of RCRA monitoring).

The highest technetium-99 concentrations occur at the south end of the WMA. Upward trends, however, are evident in the S tank farm area (well 299-W22-44, Figure 3.3) and the north end of SX tank farm (well 299-W22-45, Figure 3.3). Also, new well 299-W22-48, located toward the south end of S tank farm, intercepted an apparent plume from that area. While concentrations are low in well 299-W22-44, the sharp upward trend suggests the recent arrival of a groundwater plume. Valve pits and tank S-104, the only tank designated as leaking in S tank farm, are upgradient from wells 299-W22-44 and 299-W22-48. In the past, elevated technetium-99 has also occurred in old wells 299-W23-1, 299-W23-2, and 299-W23-7 but not in well 299-W23-3 in the southeast corner of the SX tank farm. All of the older wells inside the WMA are nearly dry. However, one last sample was obtained from 299-W23-1 and 299-W23-2 in June 2000. Well 299-W23-1 was sampled with a bailer (water could not be pumped to the surface and may not be representative) and well 299-W23-2 was sampled with a temporary pump after purging.

Technetium-99 at relatively low concentrations occurs in upgradient non-RCRA wells near the 216-S-4 and 216-S-25 cribs as well as in the two upgradient RCRA wells. Sources of the low technetium-99 concentrations are attributed to past-practice discharges of evaporator condensate and pump-and-treat water containing nitrate, technetium-99, and uranium (216-S-25 crib). These crib sources also account for the highest tritium and elevated nitrate occurrences in upgradient wells. Because these constituents (nitrate, technetium-99, and tritium) are also the primary mobile tank waste constituents, the concentrations in downgradient monitoring wells can be confused with past-practice upgradient sources. The use of constituent ratios was proposed to allow distinction between crib and tank sources (Johnson and Chou 1998; Hodges 1998; Hodges and Chou 2000).

The spatial variations in technetium-99/nitrate ratios are shown in Figure 3.4. The individual tank waste ratios from Agnew (1997) are included for comparison with the groundwater results. The upgradient wells have very low ratios compared to tank ratios. The ratios for downgradient wells (with the elevated technetium-99) are much higher than the upgradient ratios and in some cases approach the Agnew tank ratios. At the south end of the SX tank farm, the downgradient wells (299-W23-19, 299-W22-46, and 299-W22-50) all have ratios of around 0.1, which is close to Agnew's estimate for tank SX-115. This suggests there may be a common source for all three wells (e.g., the vadose zone

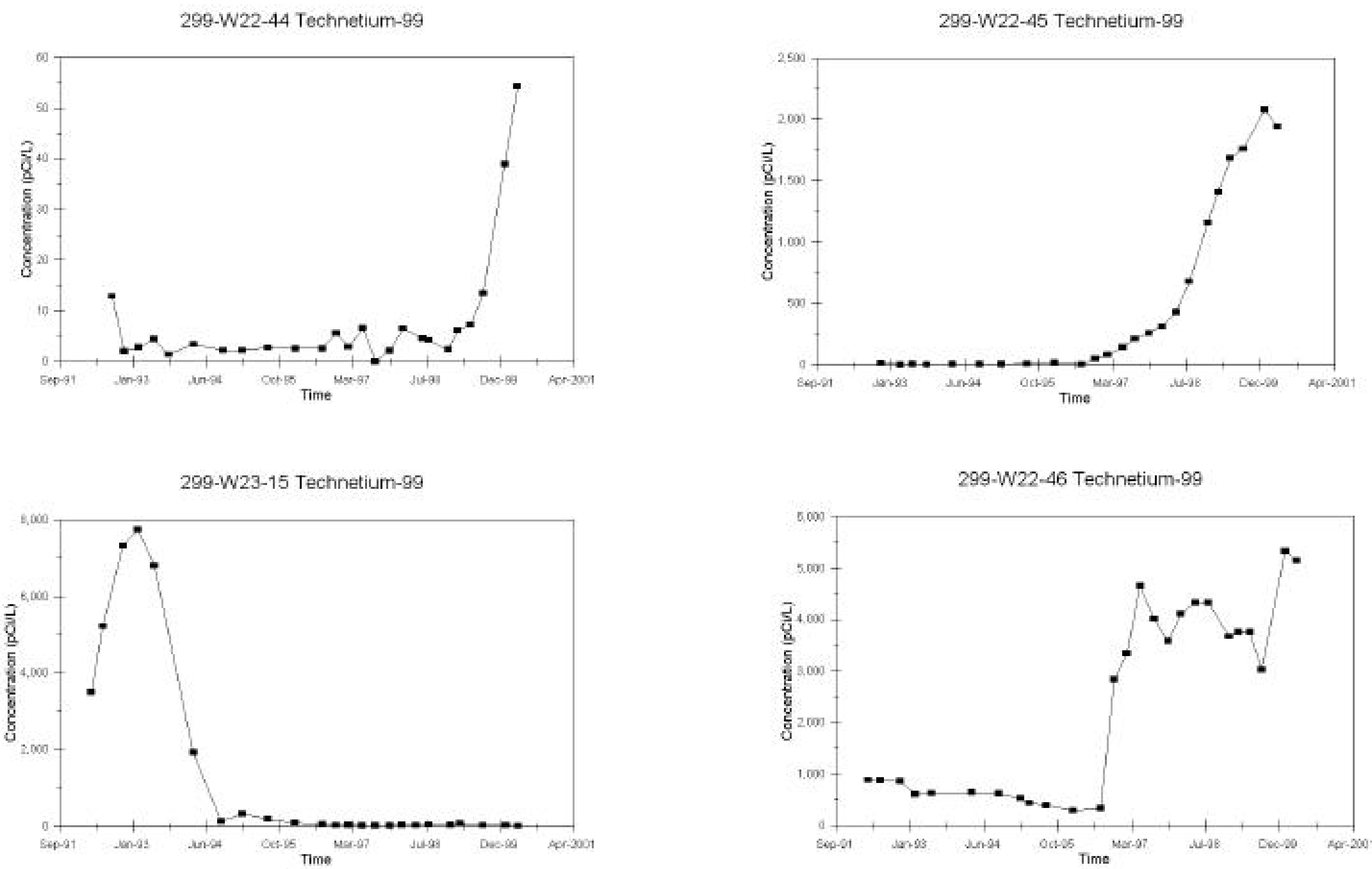

Figure 3.4. Technetium-99/Nitrate Ratios for WMA S-SX Network Wells

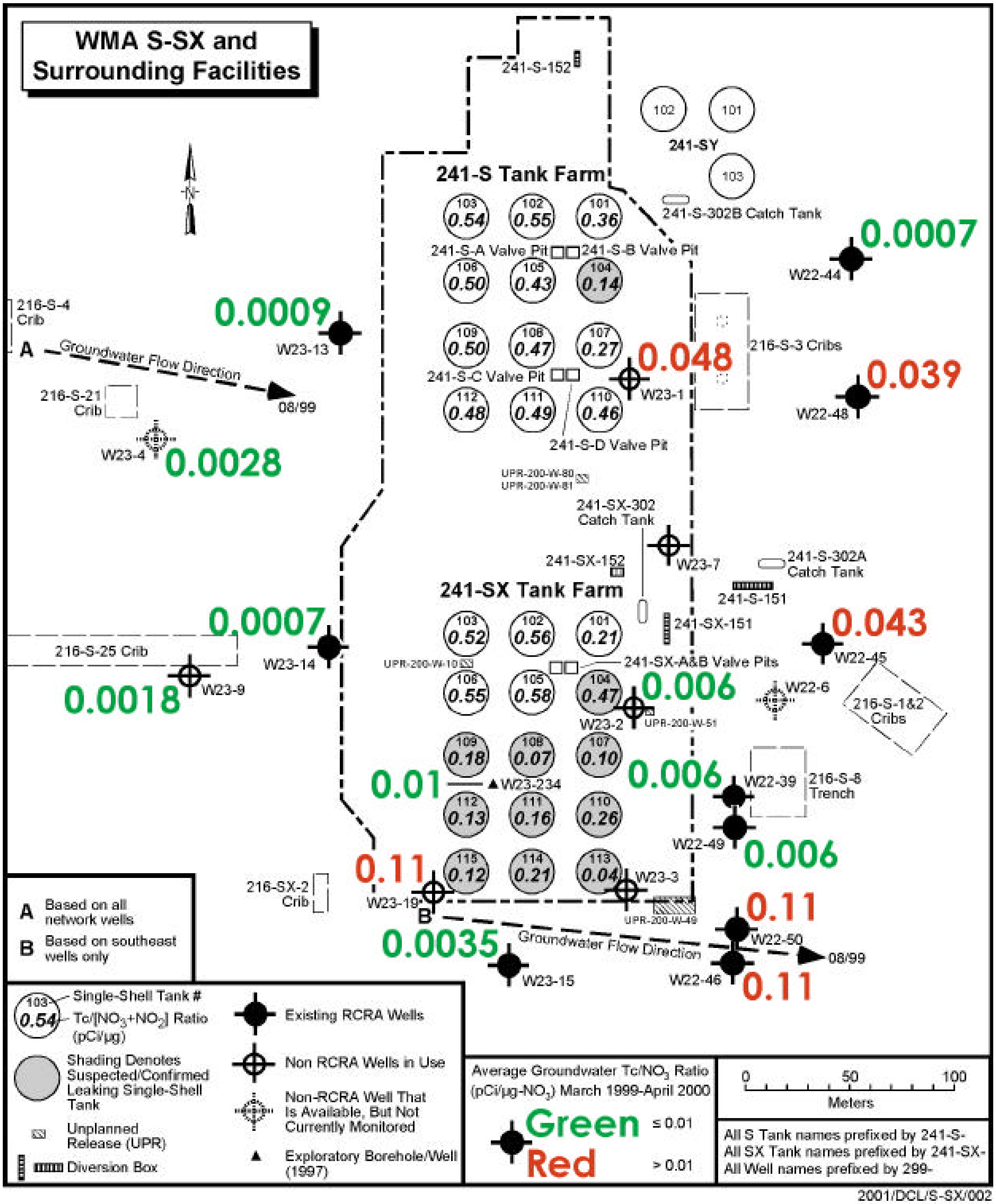

Figure 3.4. Technetium-99/Nitrate Ratios for WMA S-SX Network Wells

contamination near tank SX-115). The ratios for downgradient wells at S tank farm (299-W23-1, 299-W23-45, and 299-W23-48) are higher than upgradient ratios but lower than any of the tanks in the S farms. The S-104 tank (the only leaking tank identified in the S tank farm) has the lowest ratio of the tanks in this farm. One possibility is that upgradient nitrate has co-mingled with tank waste from S tank farm and, thus, lowered the technetium-99/ nitrate ratio in downgradient groundwater. Nevertheless, the ratios indicate there is a zone of groundwater contamination that has spread from the S tank farm area in the east to southeast direction. The extent of this contamination will be better defined with planned new wells.

Tritium/technetium-99 ratios are shown in Figure 3.5. This plot shows a dramatic difference between the upgradient source from the 216-S-25 crib and the downgradient wells with the highest technetium-99. The high upgradient ratios are consistent with the nature of the waste discharged to the 216-S-25 crib. For example, the most recent waste input to this crib was evaporator condensate from the S and SX tank farms, which was highly enriched in tritium compared to other waste constituents. Thus, the high ratio of tritium to technetium-99 is a unique characteristic of this source. Liquid waste leaking from the single-shell tanks or associated piping, on the other hand, is enriched in technetium-99 relative to the tritium. Based on Agnew's (1997) estimates of tank contents, a tritium/technetium-99 ratio of 1 to 2.6 (see Figure 3.5) would be expected. The tritium/technetium-99 ratio for groundwater with the highest technetium-99 concentration (well 299-W23-19) is similar to the SX tank waste ratios. Intermediate ratios could be a result of mixing of upgradient sources with tank waste. This may explain the intermediate ratios in the range of 100 to 500 in some downgradient wells.

Both contaminant ratios and concentrations in existing wells suggest at least two different source areas in WMA S-SX: one from the S tank farm area and one in the southern portion of SX tank farm. The S tank farm source appears to be more widely distributed but with much lower concentrations than the SX plume. Data obtained from planned new wells for fiscal year 2000 and 2001 should help to delineate the areal extent.

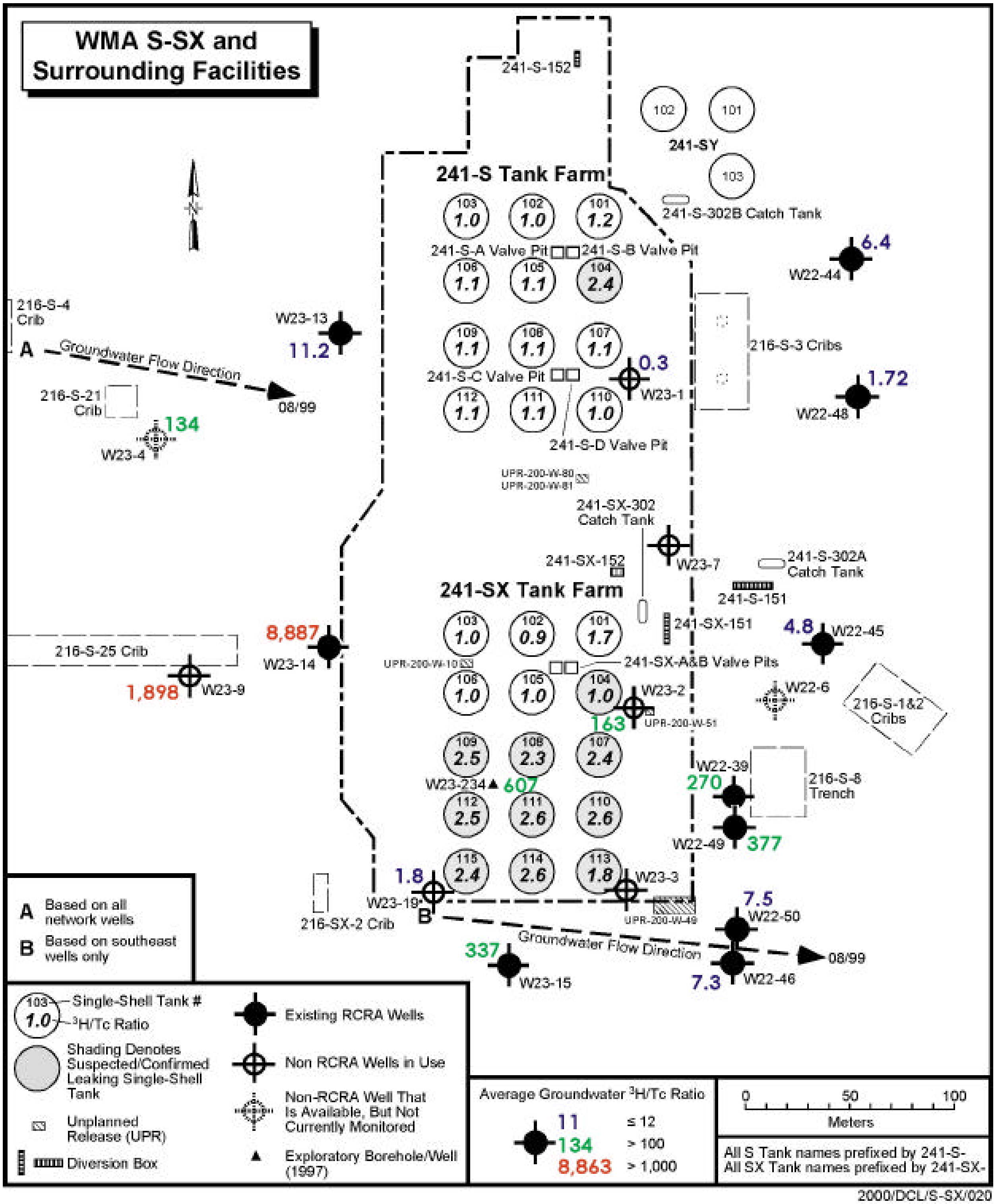

Figure 3.5. Tritium/Technetium-99 Ratios for WMA S-SX Network Wells

4.0 Monitoring Well Network Evaluation

This section provides results of computer modeling used as one tool to evaluate the theoretical spatial coverage provided by the existing well network and to optimize well coverage under different groundwater flow scenarios. Predictions of areal extent of a contaminant plume under site-specific conditions and assumptions are also performed with the same numerical model, described in the following sections. Hydrologic data summarized in Chapter 2.0 was used as input to the model.

4.1 Model Description

The model is an analytical Monitoring Efficiency Model, referred to as MEMO, which was developed to assist in design of monitoring well networks (Wilson et al. 1992). The model uses a two-dimensional plume generation routine to compute the size and shape of a plume from hypothetical source locations uniformly distributed within a source area (i.e., waste management area). The model assumes the contaminant is released as a continuous line source to a uniform or homogeneous aquifer. If a contaminant occurrence is more of a short-term transient event, then there is likelihood to overstate the computed monitoring efficiency because less lateral spreading will occur than with a continuous release source.

Major input parameters include groundwater flow direction, longitudinal and transverse dispersivities, velocity, buffer zone, and well locations. The X-Y coordinates are entered to define well locations, the waste management area boundary, and the buffer zone. The buffer zone is used to allow the hypothetical plume to expand to some point beyond the source area boundary. The farther away the buffer boundary is set, the greater the lateral spreading that will occur in the vicinity of the line of compliance where the wells are located. Thus, there is a trade off between number of wells needed to eliminate areas of non-coverage and the elapsed time when a contaminant plume would be detected. With a narrow buffer zone (boundary set close to the well locations) detection of hypothetical contaminant plumes would occur earlier but requires more wells.

4.2 Homogeneous Aquifer Simulations (Uniform Flow Field Assumption)

Longitudinal and transverse dispersivities, the parameters that control the extent of plume spreading, were previously determined using the distribution of the tritium plume in the 200 West Area.[1] These same dispersivities were used for WMA S-SX because the aquifer beneath both the northern and southern

[1] Golder Associates. 1990. Low-level Waste Burial Grounds RCRA Part B Permit Application, Section 5: Groundwater Monitoring, 903-1201, Prepared for SAIC, Richland, Washington, by Golder Associates Inc., Redmond (Seattle), Washington.

part of the 200 West Area is in the same hydrogeologic unit. Other input parameters and the values used for the computer iterations are defined below:

- X-Y coordinates: State Plane, meters.

- C_D/C_0: Dilution contour where C_D is the detection standard selected as the limiting concentration to be detected by a monitoring well, and C_0 is the source concentration in groundwater at the location of origin within the WMA. To provide adequate early warning of a release, the model should be based on a dilution contour for the more mobile potential contaminants at the site. For the S-SX, a detection limit of 10 pCi/L for technetium-99 is used as the detection standard (C_D) and 10,000 pCi/L is used as the source concentration (C_0), resulting a dilution contour of (C_D/C_0) = (10 pCi/L)/(10,000 pCi/L) = 0.001. This is a reasonable order of magnitude approximation of likely conditions at WMA S-SX.

- ldisp. Longitudinal dispersivity, meters. A value of 8.5 meters was used based on tritium plume dimensions in the 200 West Area (see Golder Associates 1990, page 102).

- tdisp. Transverse dispersivity, meters. A value of 2.5 meters was used based on tritium plume dimensions in the 200 West Area (see Golder Associates 1990, page 102).

- diffc. Effective molecular diffusion coefficient (insignificant for this application so set to zero).

- source width, meters. The length in meters of the initial source dimension (modeled as a line source of the same length spaced evenly over the entire source area). A line source length of 6 meters was used. Although larger widths might be justified to simulate a large tank leak source, the 6-meter width is considered to be conservative (i.e., a source that starts with a wider cross section will spread to a greater width at the point of compliance and would result in fewer wells needed to provide full coverage).

- lmb. First order radioactive decay constant. This term was set to zero because decay is negligible.

- cvel. Average contaminant velocity, meters/day (m/d). A value of 0.1 m/d was used for computational purposes.

Output of the MEMO model using the existing usable network is shown in Figure 4.1. The shaded areas indicate hypothetical source areas that would not be detected by the network. Figure 4.2 shows the effect of adding additional wells in the locations shown. With the added wells the theoretical detection efficiency is 100%. A downgradient buffer boundary (east and south sides) width of about 20 meters downgradient of the line of compliance was chosen. Assuming a flow rate of 50 meters per year, this represents about 5 months of additional travel time for the plume. Flow direction, as inferred from the most recent water table elevations (Chapter 2.0) is almost due east. The specific coordinates for the wells and WMA and other input values used for Figure 4.1 are in the PNNL Groundwater Monitoring Project files.

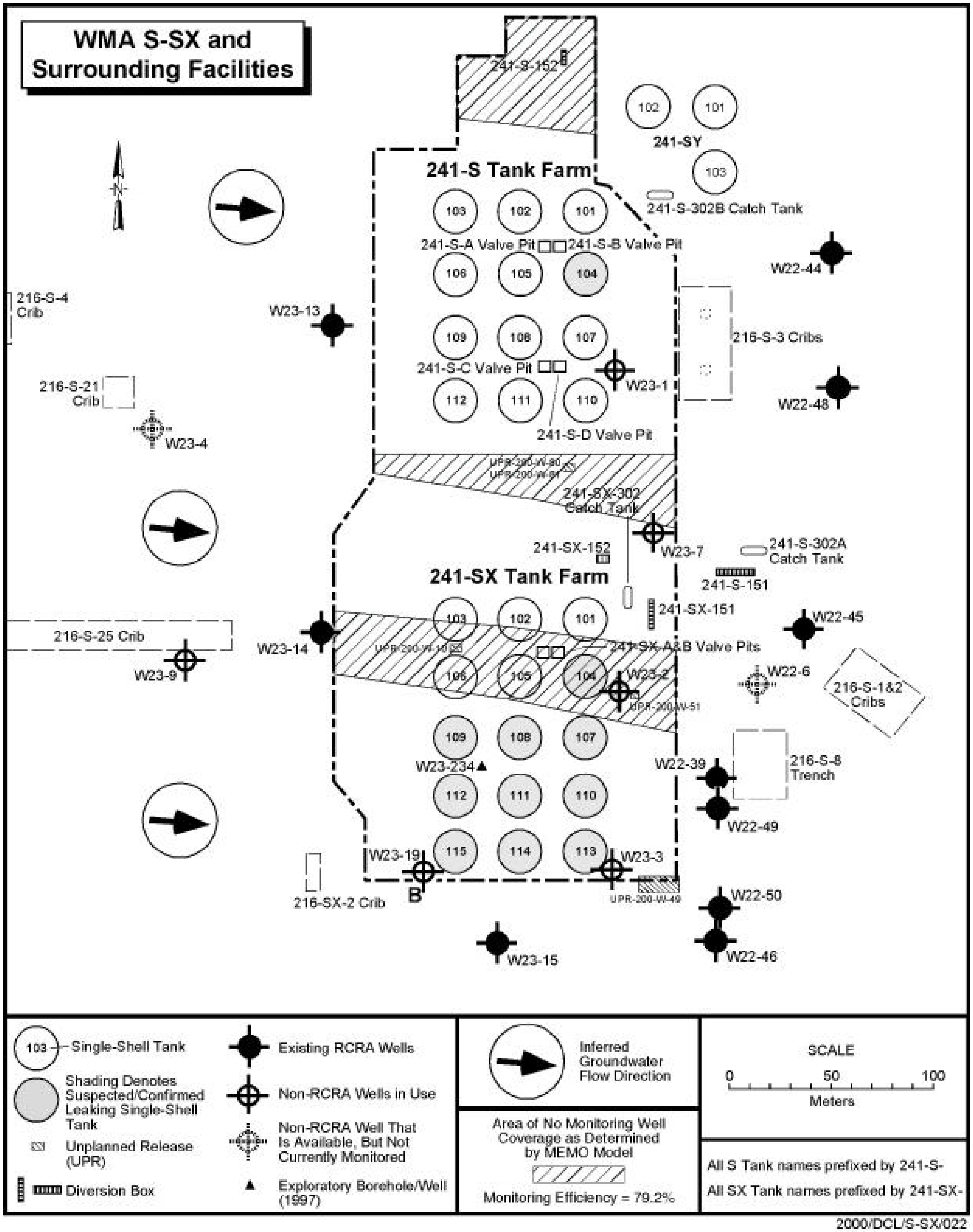

Figure 4.1. Well Detection Efficiency for Existing Wells

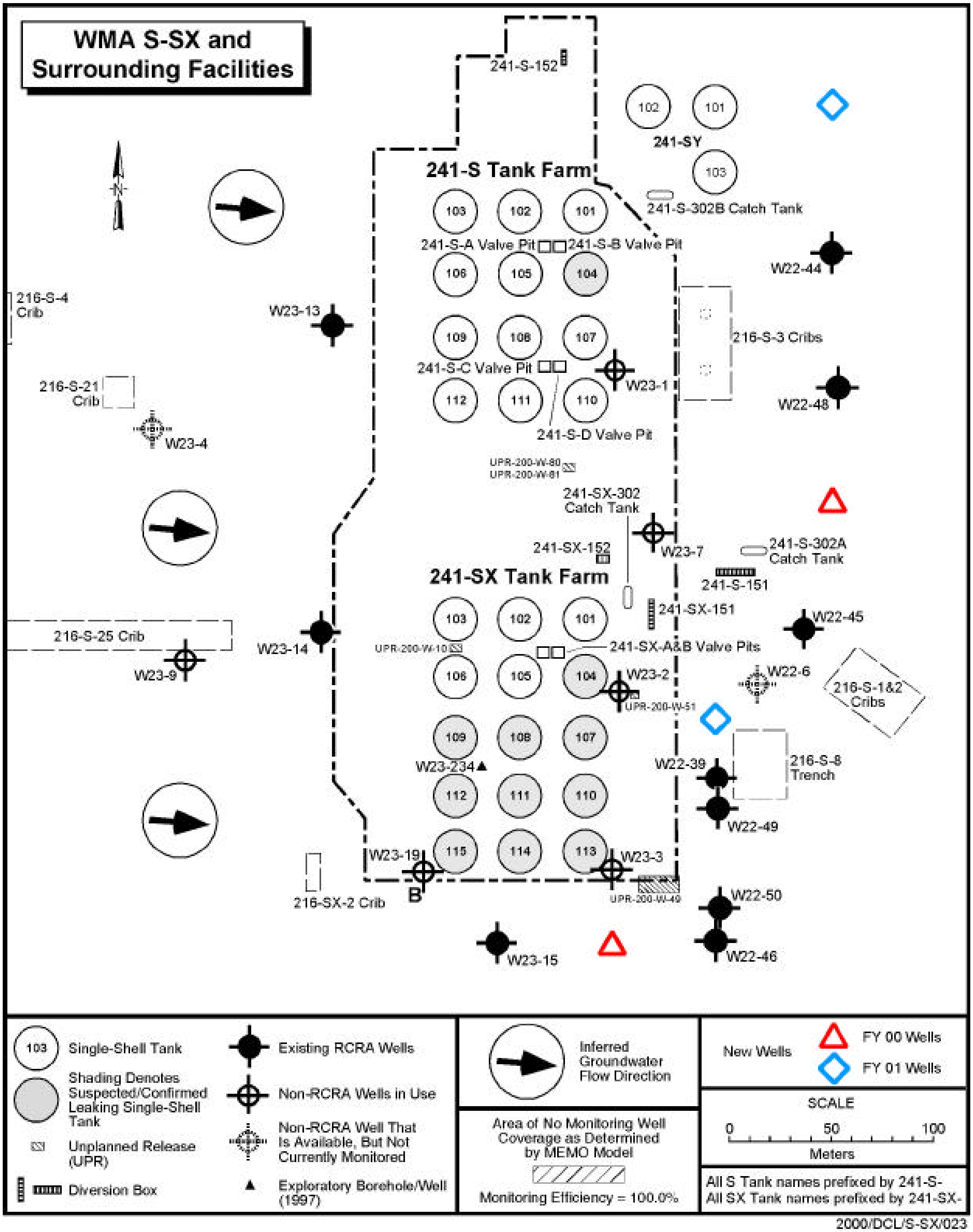

Figure 4.2. Well Detection Efficiency with Added Wells

4.3 Effect of Perturbed Flow (Non-Homogeneous Aquifer – Preferential Flow Case)

As discussed in Chapter 2.0, the non-homogeneous nature of the Ringold Formation in the study area could result in deviations from the theoretical flow directions inferred from water-table gradient analysis. The effect of impermeable zones (that may occur randomly in the aquifer host rock) would be to divert flow from the general eastward movement around the resistant features in the aquifer. At any given point, such a feature could drive the local flow of groundwater either toward the north or south. To simulate the effect of a shift in the direction of groundwater flow from the predicted easterly direction, the case shown above in Figure 4.2 was re-run with all input variables held constant except for the flow direction. The result for a southerly direction is shown in Figure 4.3. While a northerly shift could also theoretically occur because of the large scale indicators of flow direction based on the tritium plume and water table (Figure 2.1), only the southerly component was evaluated.

The results displayed in Figures 4.2 and 4.3 show the theoretical detection efficiency of the existing network plus planned new wells for fiscal years 2000 and 2001. The locations were selected by shifting well locations to minimize non-coverage (shaded area). The array suggests that the well network shown would still capture the hypothetical plume even if a contaminant originating from a source within the WMA initially started out from the west and traveled in a west-to-east direction, then was forced to turn to the right and exit along the southeast or south side of the WMA (see Figure 4.3). A wider initial plume width (more than the 6 meters used) would create wider plumes and increase the likelihood of detection and, therefore, require fewer wells. However, the dispersion parameters used could over estimate the transverse (width) spreading, offsetting the narrow source width chosen. The buffer zone was set close to the line of compliance wells, which also requires more wells for the same coverage (efficiency) than if the buffer zone were set at a greater distance away. Even in the case of southerly flow direction, the area of non-coverage is actually located along the western side of the WMA where there are few sources of contamination or leaking waste tanks. Efficiency for a southeasterly flow (not shown) was 99%.

4.4 Predicted Plume Dimensions

Some idea of plume dimensions can be gained using simple two-dimensional dispersion models. A subroutine of the MEMO model used for evaluation of monitoring well detection efficiency and well spacing can be used for this purpose. Application of the groundwater contaminant plume transport model (PLUME, a subroutine of MEMO) requires some simplifying assumptions and a conceptual model. A single source scenario is described in the following paragraphs. The single source case is the basis for the hypothetical plume shown in Figure 4.4. New data from either the vadose zone study or from new groundwater monitoring wells may result in revised conceptualizations.

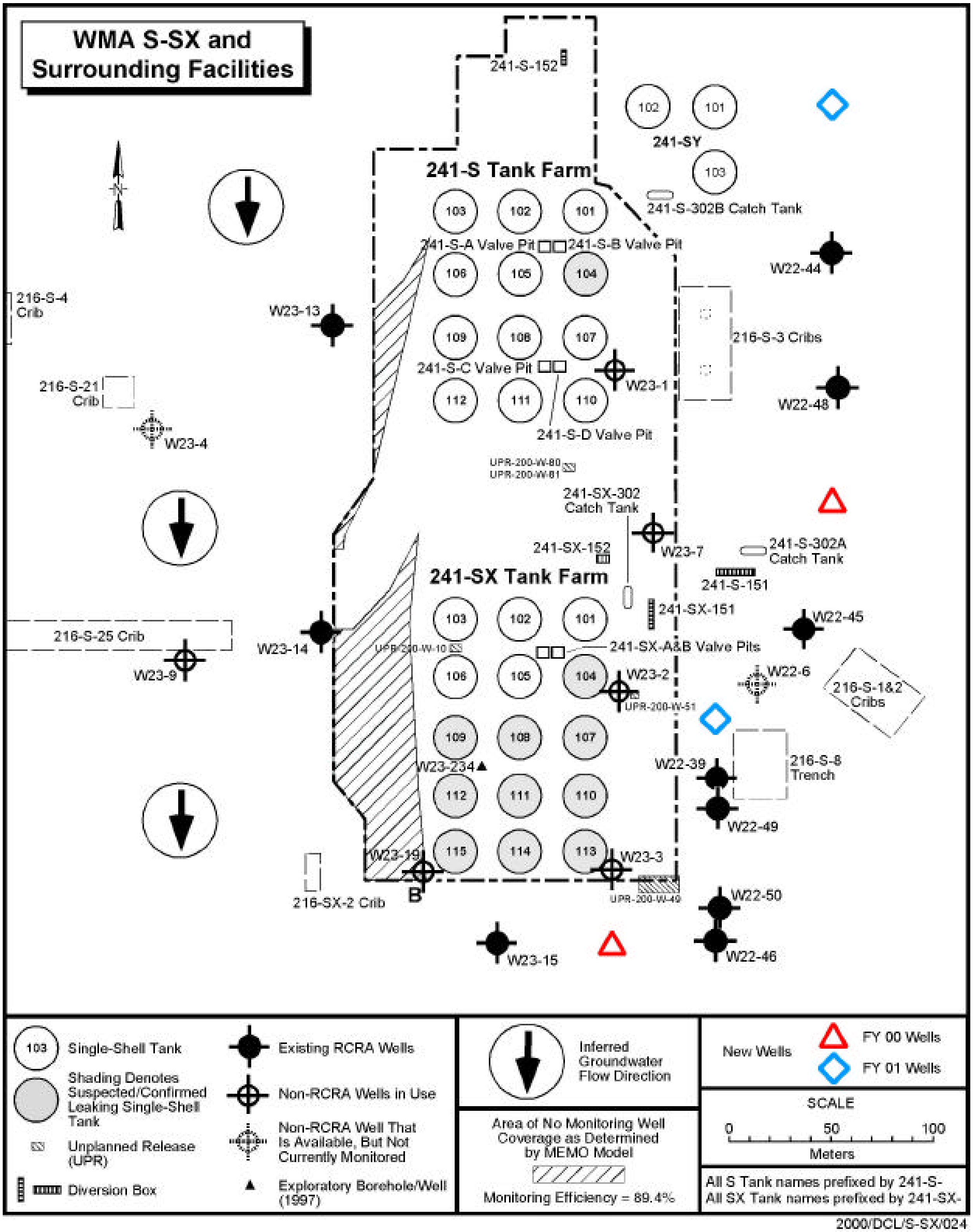

Figure 4.3. The Effect of Flow Direction on Well Detection Efficiency

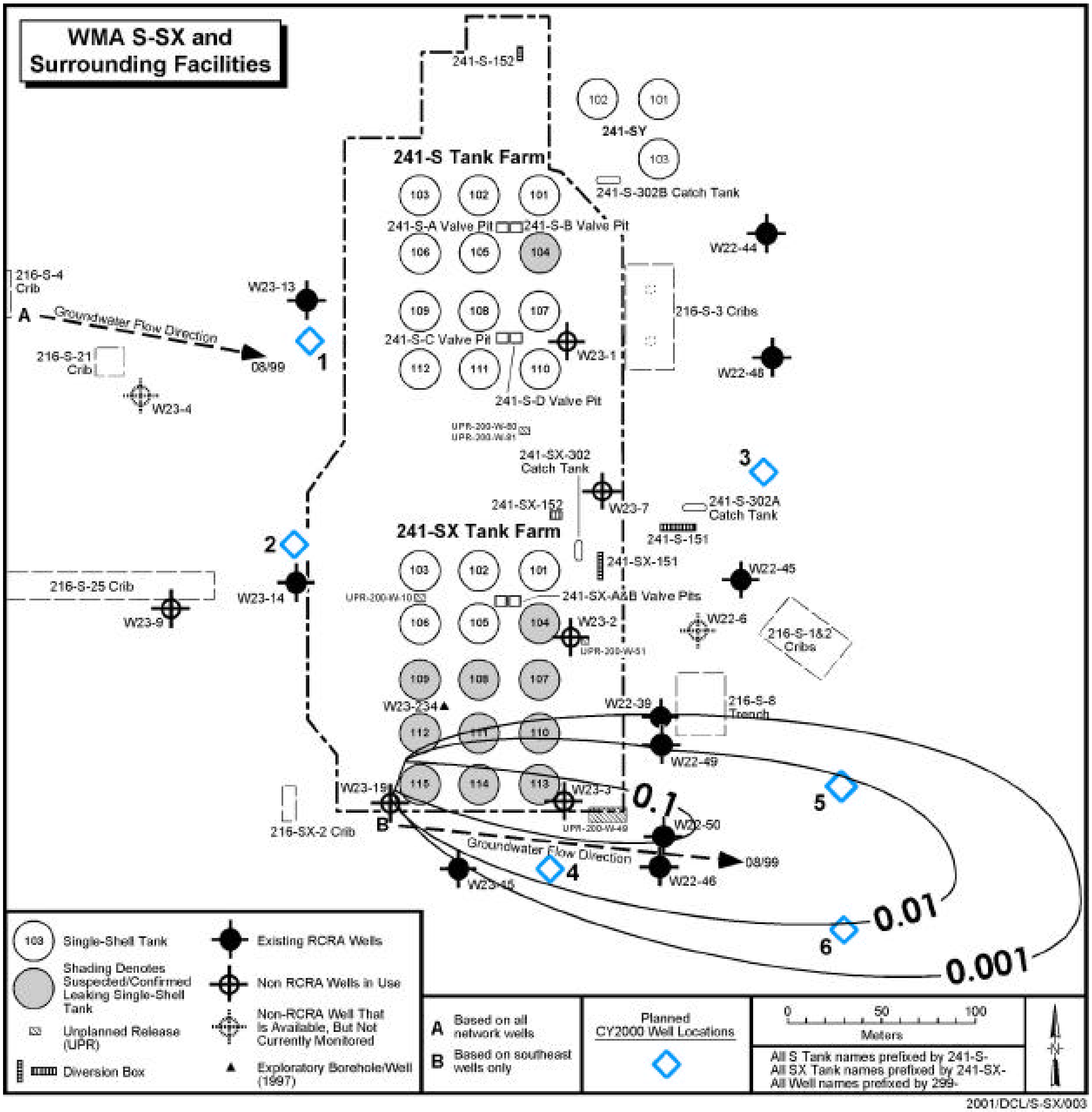

Figure 4.4. Theoretical Groundwater Plume Dispersion Pattern from a Tank Leak Source in the SX Tank Farm (MEMO model)

4.4.1 Conceptual Model and Assumptions (Single Source Case)

1. **Source** – The dominant or primary source is assumed to be drainage of past leakage near tank SX-115 through the vadose zone to groundwater. Release rate to groundwater is assumed to be continuous and at a constant rate. (The roughly constant technetium-99 concentration in well 299-W22-46 that has persisted for over 4 years is consistent with a continuous source). As previously discussed, the technetium-99/nitrate ratios in the wells with elevated technetium-99 (299-W22-46, 299-W22-50, 299-W23-15, and 299-W23-19) have all exhibited ratios of about 0.1, either at the present time or in the past (e.g., 299-W23-15). This suggests a common origin. The size of the initial source at the water table is assumed to be 20 meters or about the width of a single-shell tank.

2. **Flow rate and time period of interest (advection time)** – Approximate rates and time periods can be inferred from time series data available in selected wells. Technetium-99 peaked in 1993 in well 299-W23-15. The flow direction at that time was east-southeast. It was assumed that as groundwater flow shifted to a more easterly direction, the plume no longer was intersected by this well (299-W23-15). The plume then migrated to the next nearest downgradient monitoring well (299-W22-46, see Figure 3.3, bottom two plots). Based on the distance (100 meters) between these two wells and elapsed time (January 1992 to April 1996 or 4.3 years), a flow rate of about 0.06 meter per day is implied. (Preliminary hydraulic test results for these wells suggests very low flow rates that are consistent with this estimate; see Appendix A). Assuming 0.06 meter per day is representative, and taking the distance between tank SX-115 and well 299-W23-15 (~50 meters), the estimated date when contaminants first reached groundwater and started migrating away from the tank would have been early 1990. It is assumed that the reason technetium-99 in well 299-W22-46 (see Figure 3.3) has remained relatively constant since 1996 is that a fairly continuous release to groundwater from the vadose zone occurred (and continues to occur) near or in the vicinity of tank SX-115. The above considerations suggest a time period of approximately 10 years has elapsed since breakthrough to groundwater first began beneath tank SX-115. Thus, 10 years was used for the run (or advection) time in the MEMO simulation.

3. **Aquifer conditions** – The aquifer is assumed to be homogeneous for purposes of the simulation. The likely deviation from the ideal would involve “fingering” of the plume. It is also assumed that the contaminant plume remains restricted to the upper few meters of the aquifer. This is consistent with the downgradient depth profile for tank waste constituents (see Figure 3.2).

4. **Initial concentration** – The concentration at the origin, C_o, is assumed to be close to the concentrations detected in new well 299-W23-19, located immediately adjacent to tank SX-115. The average technetium-99 concentration for this well is 50,750 pCi/L (see Table 3.1).

5. **Contour intervals (C/C_0)** – Intervals of 0.1, 0.01, and 0.001 were chosen for the simulation. If the above average represents C_o, then the 0.1, 0.01, and 0.001 contours would be 5,000, 500, and 50pCi/L respectively.

6. **Dispersivities** – A transverse dispersivity of 1.2 meters was used (or a longitudinal to transverse ratio of 7.0). A smaller transverse value was used than Golder suggested, better approximately the observed well data at the sound end of the SX tank farm.

The predicted or hypothetical plume based on the above assumptions is shown in Figure 4.4. If the simulations and assumptions approximate actual conditions, the size of the plume that would be of concern for potential remediation is relatively small. For example, the area of the hypothetical plume exceeding 10 times the drinking water standard (9,000 pCi/L remediation goal for the nearby UP-1 plume) would be less than the area bounded by the 0.1 contour shown in Figure 4.4 (perhaps 1,000 to 2,000 square meters). This area is 10 to 20 times less than the original UP-1 plume prior to the pump-and-treat operation near U Plant (see annual report, Hartman et al. 2000).

Results from new monitoring wells planned for 2000 and 2001 (see Figure 4.4) should help to define the actual extent of the plume.

5.0 Maximum Contaminant Concentrations

Table 5.1 shows the maximum concentrations detected for the primary constituents of concern identified for this assessment (Johnson and Chou 1999a) for each well included in the monitoring network for the period November 12, 1997, to April 3, 2000. Non-RCRA wells are included as well as the RCRA compliant wells in the network. Samples collected during drilling of the new wells 299-W22-48, 299-W22-49, and 299-W22-50 were also included in Table 5.1. Only filtered (0.4 µm) metal results were included in the summary. Results for anions, volatile organic compounds, and radionuclides are all based on unfiltered samples. The last column shows the highest maximum contaminant concentration (values in bold type) divided by the applicable maximum contaminant level or drinking water standard, referred to as the relative hazard index for purposes of this report.

The highest relative hazard index (58) shown in Table 5.1 is for technetium-99 in well 299-W23-19 located immediately adjacent to tank SX-115 (the nature of this occurrence was previously discussed in Chapter 3.0). The second highest relative hazard index value (28) is for carbon tetrachloride in well 299-W23-15. The maximum concentration observed in this well was 140 µg/L as compared to the maximum (100 µg/L) for upgradient well 299-W23-4. The occurrence of carbon tetrachloride in this area is attributed to past discharges to the Plutonium Finishing Plant cribs and trenches and to wastewater containing dissolved carbon tetrachloride that was routed to the U Pond via the ditches from the Plutonium Finishing Plant and U plant areas (Hartman et al. 2000, page 2.113).

The third highest relative hazard index (25) is for tritium in upgradient well 299-W23-9. The source of this tritium is residual drainage from the 216-S-25 crib, which received condensate from the S evaporator in the past. The nature of this source has been discussed in previous annual groundwater reports and in the preliminary assessment report and in the subsequent assessment plan and addendum (Johnson and Chou 1998; 1999a,b).

Nitrate has the fourth highest relative hazard index that occurs in the same well with the highest technetium-99, 299-W23-19. The correspondence of the highly mobile nitrate and technetium-99 in tank waste has been discussed previously.

Other minor exceedances are for aluminum, iron, manganese, and uranium. Except for iron in RCRA well 299-W23-15, these exceedances occurred in samples collected during drilling of new wells 299-W22-48, 299-W22-49, and 299-W22-50. The cause of the elevated iron and manganese in these samples is unknown. One possibility for the high iron is that an iron rich bentonite may have fallen into the screened interval during well installation. The fine fraction of bentonite is a smectite clay that is often enriched in iron (~4 to 30 weight percent iron). The particle size of the smectite clay fraction is small enough to pass through 0.4 µm filters. If this is the case, this sub micron colloidal phase could pass through the filter and would be dissolved in the acidified sample. Thus, iron would be released into solution. Some limited laboratory testing is planned to evaluate the feasibility of this possible explanation. Other metals, such as aluminum or manganese, may become artificially elevated by this mechanism as well.

Table 5.1. Maximum Contaminant Concentrations for Groundwater Samples Collected from WMA S-SX Network Wells (November 1997 to April 2000)

Analyte	MCL	W22-39	W22-44	W22-45	W22-46	W22-48	W22-49	W22-50	W23-1	W23-2
Chromium [(a)] (μg/L)	100	16.3	5[(b)]	27.4[(b)]	26	7.1	4.6U	10.4	29	6.1
^{99}Tc (pCi/L)	900	120	56	2,080	5,330	720	58.3	4,240	2,890	75.6
Nitrate (as NO_3) (μg/L)	45,000	16,822	37,628	47,367	49,580	18,593	13,546	57,991	50,023	11,952
Uranium (μg/L)	20	4.81	6.48	7.97	5.91	3.23	3.27	**30.9**	8.06	6.57
Gross alpha (pCi/L)	15	3.56	4.81	6.36	4.36	1.21	2.24	20.9	5.68	4.65
Gross beta (pCi/L)	50	37.5	20.7	768	1,836	223	22	1,420	1,090	29.3
Tritium (pCi/L)	20,000	27,100	238U	18,800	58,700	249U	22,000	31,400	1,010	12,700
^{90}Sr (pCi/L)	8	0.261U	0.295U	1.4U	4.92U	0U	0U	1.03U	**7.54**	0.205U
^{137}Cs (pCi/L)	200	1.82U	2.42U	2.78U	4.59U	0U	1.7U	0U	1.82U	0U
Iron[(a)] (μg/L)	300	118	146[(b)]	65.9	122	54.9	85.8[(b)]	95.5[(b)]	59	39.5
Manganese[(a)] (μg/L)	50	7.7	8.1[(b)]	3.8	4.3	**306**	244	167	7	11.8
Carbon tetrachloride (μg/L)	5	3.8	6.2	12	30	4	6	19	--	--
Fluoride (μg/L)	4,000	473	400	480	440	490	550	**730**	458	424
Aluminum [(a)] (μg/L)	50	51	58.3U	41.8U	45.7	41.8U	64.6	**93.2**	31.3U	20.6U
pH	[6.5, 8.5]	[7.93, 8.48]	[7.92, 8.25]	[8.07, 8.35]	[7.89, 8.33]	[7.97, 8.59]	[8.1, **9.09**]	[8.1, 8.14]	[**7.33**, 8.19]	[8.11, 8.26]

Analyte	W23-4[(c)]	W23-9[(c)]	W23-13[(c)]	W23-14[(c)]	W23-15	W23-19	W23-234	Max[(d)]	Max/MCL
Chromium [(a)] (μg/L)	4.6U	8.5	33.9	12.5	8.1[(b)]	**89.8**	7.5	89.8	0.9
^{99}Tc (pCi/L)	21.1	408	10.6U	218	72.1	**52,300**	80.4	52,300	58.1
Nitrate (as NO_3) (μg/L)	4,869	165,562	7,698	134,575	14,697	**562,204**	19,080	562,204	12.5
Uranium (μg/L)	24.4	25.8	16.5	18	14.6	17.6	3.4	30.9	1.5
Gross alpha (pCi/L)	14.5	17.3	11.7	9.82	10.8	**21.9**	--	21.9	NA
Gross beta (pCi/L)	15	56.6	11.4	19.7	27.8	**23,000**	--	23,000	NA
Tritium (pCi/L)	1,540	**502,000**	215U	382,000	22,200	95,800	138,000	502,000	25.1
^{90}Sr (pCi/L)	0.222U	0.481U	0.189U	0.425U	0.36U	9.63U	0.2U	7.54	0.94
^{137}Cs (pCi/L)	3.96U	0U	1.33U	1.71U	2.31U	1.63U	1.3U	Not detected	NA
Iron[(a)] (μg/L)	81.5	83.3	154	110	**938**	46	48.2	938	3.1
Manganese[(a)] (μg/L)	2.4	30.3	10.2	12.7	20.1	203	5.7	306	6.1
Carbon tetrachloride (μg/L)	100	2	11	0.51	**140**	22	--	140	28
Fluoride (μg/L)	340	319	390	350	490	340	497	730	0.2
Aluminum [(a)] (μg/L)	41.8U	41.8U	41.8U	33.5U	83.2	41.8U	33.5U	93.2	1.9
pH	[8.0, 8.09]	[7.72, 8.1]	[7.72, 8.58]	[7.78, 8.52]	[7.74, 8.12]	8.05	8.54	[7.33, 9.09]	NA

Note: All well numbers prefixed by 299-. U denotes analytical result is not detected. **Bold** indicates well with maximum.
(a) Filtered sample results.
(b) Outliers removed.
(c) Upgradient wells.
(d) Maximum across all network wells.

The moderately elevated uranium concentration was collected from below (see Table 3.2) the lower mud unit at well 299-W22-50. The origin of elevated uranium is uncertain, but is most likely related to a natural source of uranium. A large fraction of the Ringold unit below the mud layer is granitic in origin. Volcanic ash beds enriched in uranium also occur in the drainage basin (Milne 1979). Uranium may leach more readily from this type of silicate matrix. Elevated uranium has been reported in groundwater elsewhere in the Columbia Basin (Ichimura and Crosby 1981) and was attributed to leaching of adjacent granitic source rock.

Hexavalent chromium (Cr^{6+}) was analyzed in selected wells to confirm that the chromium identified by analysis of filtered water samples using inductively coupled plasma (ICP) spectroscopy was actually hexavalent chromium (the chemical form on which the maximum contaminant level is based). Unfiltered water samples from wells 299-W22-45, 299-W22-46, and 299-W23-15 were analyzed using a HACH spectrophotometric method that is specific for hexavalent chromium. Results of this comparison (six sample sets) indicated most of the chromium present was hexavalent chromium and that Cr^{6+} could account for the chromium analyzed by the ICP method on filtered sample splits. These results were discussed previously (Hartman et al. 2000, page 2.143); however, some discrepancies occurred in subsequent results that are most likely analytical in nature. Some additional comparison checks may be appropriate.

Other constituents of concern not listed in the table were analyzed in selected wells with the highest likelihood of occurrence. For example, groundwater samples from well 299-W23-234 near tank SX 108 and well 299-W23-19 near tank SX-115 were analyzed for iodine-129, neptunium-237, plutonium-238, -239, and -240, and americium-241 (all unfiltered samples). The vendor reported non-detect results for all these constituents.

One older well (299-W23-7) not included in the Table 5.1 summary has been discussed previously in both annual groundwater reports and in the assessment plan. This well no longer produces water and has been removed from the sampling list and placed on the well decommissioning list. The occurrence of cesium-137 (particulate) of up to 47 pCi/L in this well may be a result of fall-in from work done on the well in the past (Johnson and Chou 1999b).

6.0 Conclusions

Additional assessment characterization and monitoring activities were conducted to further evaluate the rate, extent, and concentration of contaminants in groundwater beneath WMA S-SX. Installation of additional groundwater monitoring wells, hydrologic testing, and sampling and analysis (both vertically and areally) also provided new information, which resulted in the following conclusions.

6.1 Rate and Extent of Contaminants

Depth distribution data near one of the downgradient contaminant occurrences (299-W22-46 and 299-W22-50) indicated that most of the mobile contaminants from tank waste at that location were at the very top of the aquifer (upper 5 meters).

The areal extent of contamination beneath the WMA is irregular. This in part may be due to inadequate monitoring well coverage. Nevertheless, indications from the new wells and resampling of existing wells, both within the tank farm and adjacent to it, are that the most significant groundwater contamination attributable to WMA S-SX originates in the SX tank farm area.

Hydrologic testing was conducted on newly installed wells and on selected existing wells. Associated hydraulic conductivities from both drawdown and slug tests, and effective porosities from selected tracer tests (borehole dilution tests), were determined for this assessment. Based on calculated rates and direct observations (tracers and contaminant plume arrival times) a contaminant plume originating in WMA S-SX should travel very slowly (30 to 50 meters per year) in the vicinity of the WMA. Because the contaminant increase in 299-W22-46 first appeared in that well in late 1996, this plume is estimated to have spread downgradient about 100 to 150 meters to the east-southeast of the line of compliance wells. Two new midfield wells to be drilled in 2000 and 2001 are intended to evaluate the spread of the contaminant plume in this downgradient direction.

6.2 Concentration of Contaminants

The highest concentration of contamination found, relative to a maximum contaminant level or drinking water standard during the report period, was 52,300 pCi/L of technetium-99, which is about 58 times the drinking water standard of 900 pCi/L. (The June 2000 result was 63,700 pCi/L.) At the present time, this zone of elevated technetium-99 appears to be localized. Overall, the extent of the contaminant plume attributable to WMA S-SX appears to be small compared to the large technetium-99/uranium plume at the nearby UP-1 site. Whether or not remediation of groundwater contamination near tank SX-115 (well 299-W23-19) would be effective is beyond the scope of this report. However, if this contamination is due to slowly migrating tank waste that has just reached the water table, and if previous vadose transport modeling results (Ward et al. 1997) are relevant at this single-shell tank site, very slow drainage to groundwater can be expected. If this is true, groundwater remediation efforts would be of little value unless the driving force for downward vadose zone transport at this location is eliminated or the technetium-99 is immobilized.

6.3 Well Network

The optimized network design (location and number of existing and planned new wells) should provide adequate spatial coverage for detection of contaminants from the WMA during both the operational and post closure period. However, additional consideration of extended sampling depths may be needed. Also, if vadose zone transport modeling that has been done to date is representative, then the vadose zone near tank SX-115 may "bleed" very slowly for tens of years before a significant decline occurs. Further declines in the water table may require deepening or replacement of the sampling wells in the future to intersect a more deeply distributed plume.

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Appendix A

Preliminary Analysis of Hydrologic Testing Results

Battelle
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Project No. F05158

Internal Distribution

C.J. Chou
F.N. Hodges
S.P. Luttrell
D.R. Newcomer
P.D. Thorne
PFile/LB

Date September 26, 2000

To V.G. Johnson

From F.A. Spane

Subject Preliminary Results for FY-99 and FY-00 Detailed Hydrologic Characterization Tests Conducted in the WMA S-SX, TX-TY, and T

This letter report presents preliminary results obtained from detailed hydrologic characterization tests conducted within the WMA S-SX, TX-TY, and T during FY-99 and FY-00. These results are in the process of being formally documented in several PNNL technical reports (e.g., Spane et al. 2000). This letter report is being issued as an interim measure to meet current hydrologic data needs of various WMA projects, prior to formal technical report issuance. The letter report only provides the preliminary results for the various detailed hydrologic characterization test elements, and does not present discussions pertaining to test descriptions, and analytical methods and result comparison. These discussions will be presented in detailed fashion in the subsequent technical reports.

Detailed Hydrologic Characterization Program

As part of the Hanford Groundwater Monitoring Project, Pacific Northwest National Laboratory conducts detailed hydrologic characterization tests within wells at selected locations to provide information pertaining to the hydraulic properties and groundwater flow characteristics of the unconfined aquifer. The following identifies and briefly describes the various characterization components employed in FY-99 and FY-00, as part of the detailed hydrologic characterization program. Various individual test element activities include:

Groundwater Flow Characterization: for quantitative determination of groundwater flow direction and hydraulic gradient conditions

Barometric Response Evaluation: for determining well response characteristics to barometric fluctuations; for estimating vadose zone transmission characteristics; and for removal of barometric pressure effects from hydrologic test responses

Slug Testing:	for evaluating well development conditions and to provide preliminary hydraulic property information (e.g., hydraulic conductivity) for design of subsequent hydrologic tests
Tracer-Dilution Test:	for determining the vertical distribution of hydraulic conductivity and/or groundwater flow velocity within the well-screen section, and for identifying vertical flow conditions within the well column
Tracer-Pumpback Test:	for tracer removal and characterizing effective porosity, an important hydraulic transport parameter
Constant-Rate Pumping Test:	conducted in concert with tracer-pumpback phase. Analysis of drawdown and recovery data provides quantitative, large-scale hydraulic characterization property information, e.g., hydraulic conductivity, storativity, specific yield
Step-Drawdown Test:	for determining well efficiency and well loss for the well-screen section; for removal of well loss effects from hydrologic test response
In-Well Vertical TracerTest:	for determining the existence of vertical flow within the well-screen section

Accurate delineation of the prevailing groundwater-flow direction and hydraulic gradient, I, conditions is critical for proper evaluation of groundwater contaminant movement. Within study areas of small size and/or having low gradient conditions, detailed groundwater flow characterization can be difficult. A method that facilitates groundwater flow characterization in such areas is the use of trend-surface analysis of representative monitoring well total head measurements (not well water-level elevation). A description of the use of trend-surface analysis for detailed characterization of groundwater flow conditions is presented in Spane (1999).

Slug testing is designed primarily to provide initial estimates of hydraulic conductivity, K, for the design of subsequent, more quantitative hydrologic tests. At each well, slug tests are conducted using at least two different stress levels to provide information pertaining to well development and possible presence of near-well heterogeneities. A detailed description of the design, performance and analysis of slug test characterizations is presented in Butler et al. (1994) and Butler (1997).

Tracer dilution and tracer pumpack/constant-rate pumping and recovery tests are conducted at single-well sites. For the tracer-dilution test, a bromide solution of known concentration is circulated/mixed within the well-screen section. The decline of tracer concentration (i.e., "dilution") with time within the well screen is monitored directly using a vertical array of bromide specific-ion electrode probes located at known depth intervals. Based on the dilution characteristics observed, the vertical distribution (i.e., heterogeneity) of hydraulic properties and/or flow velocity can be estimated for the formation within the well-screen section. The presence of vertical flow within the well screen can also be identified from the probe/depth dilution response pattern. A description of the performance and analysis of tracer-dilution test characterization investigations is provided in Halevy et al. (1966), Hall et al. (1991), and Hall (1993).

For the tracer pumpback, a constant-rate pumping test is initiated after the average tracer concentration has decreased (i.e., diluted) to a sufficient level within the well screen (usually a 1 to 2 order of magnitude reduction from the original tracer concentration). The objective of the pumpback test is to "capture" the tracer that has moved from the well into the surrounding aquifer. Tracer recovery is monitored by measuring the tracer concentration in water pumped from the well. The time required to recover the centroid of tracer mass/concentration provides information of the aquifer effective porosity, n_e. Effective porosity is a primary hydrologic parameter controlling contaminant transport. Once estimates for n_e, K, and I have been determined, the average aquifer groundwater flow velocity, v_a, can also be calculated.

The constant-rate pumping test may be extended for a time duration longer than required for capturing the tracer centroid. The extended pumping time enables quantitative large-scale characterization of the surrounding hydraulic properties. The time required to obtain representative hydrologic property results can be determined by using diagnostic derivative analysis results of the drawdown data obtained from the pumped and nearby observation well locations. A detailed description of the use of derivative analysis techniques is provided in Spane (1993) and Spane and Wurstner (1993).

Following termination of the constant-rate pumping test phase, the recovery of water levels within the pumped well and surrounding observation wells can also be monitored. The time required for recovery monitoring can be assessed in a manner similar to drawdown data collected during the pumping phase, through the use of diagnostic derivative analysis. For general planning purposes, however, recovery monitoring should be maintained for a period equal to the pumping period and preferably longer. Analysis of the associated pressure drawdown and recovery responses at the surrounding observation wells provides the basis for determining standard, large-scale hydraulic properties within the tested aquifer. These hydraulic properties include: horizontal conductivity (K_h), transmissivity (T), storativity (S), and specific yield (S_y). In addition, detailed hydrologic property characterization obtained from compositely analyzing drawdown and recovery data from multiple observation wells include: vertical anisotropy (K_v/K_h) and horizontal anisotropy (K_{hx}/K_{hy}). The vertical and horizontal anisotropy parameters are the principal hydraulic parameters controlling the directional contaminant transport within the local area.

A group of tables is presented in this letter report that summarize the results from various detailed hydraulic characterization activities. Table 1 provides a summary of the various detailed hydraulic characterization elements. Table 2 lists the preliminary analysis results for hydraulic conductivity and transmissivity determined from slug tests and constant-rate pumping tests. Table 3 presents pertinent information pertaining to tracer-dilution testing, and estimates for lateral groundwater flow velocity within the well screen, v_w. Table 4 presents results of tracer pumpback testing and associated estimates for effective porosity, n_e, and average aquifer groundwater flow velocity, v_a. Table 5 lists the results of groundwater flow characterization (hydraulic gradient, I, and groundwater flow direction), based on trend-surface analysis, for the various well sites selected for tracer testing.

Data Discussion

<u>Table 2</u>

Table 2 presents estimates obtained from slug testing and constant-rate pumping tests. The range for K listed for slug tests represent the average K value as determined using the Bouwer and Rice method and the type-curve matching procedure. Constant-rate pumping test results include the analysis of drawdown and/or recovery data using the methods identified previously. A close correspondence in estimates for K is evident between the two test methods. It should also be noted that the test analysis was completed independently by different analysts, i.e., F.A. Spane: slug tests and P.D. Thorne: constant-rate pumping tests.

<u>Table 3</u>

Table 3 lists pertinent information pertaining to the tracer-dilution tests performed. Several wells exhibited vertical flow conditions (denoted by VF in the table), which largely invalidate the results of the test. The vertical flow conditions detected during the tracer-dilution testing (i.e., well 299-W10-26: downward; well 299-W14-13: downward; and 299-W22-49: upward) were also corroborated independently directly using electromagnetic vertical flowmeter surveys conducted at these wells, as reported in Waldrop and Pearson (2000).

It should be noted that the v_w estimates based on the tracer-dilution tests are strictly for in-well groundwater flow conditions. The relationship between v_w and aquifer groundwater flow velocity, v_a, is shown in equation (1) below:

$$v_w = v_a \, n_e \propto \qquad (1)$$

where, $\propto$ = groundwater flow distortion factor; dimensionless, common range 0.5 to 4

Average well flow velocities ranged between 0.007 to 0.311 m/d. It should be noted that the lowest average value of 0.007 m/d recorded at well 299-W22-48 (WMA S-SX), is a result of averaging depth/well velocity conditions that indicate very little flow within the lower part of the well screen. The value of 0.023 m/d indicated for the well screen maybe more reflective of actual aquifer conditions. The highest value of 0.311 m/d calculated for well 299-W15-41 (WMA T) is higher than expected, and may be the result of extraneous hydrologic effects imposed by the nearby 200-ZP-1 pump and treat facility. This well location is well within the potential radius of influence distances reported in Spane and Thorne (2000) and, therefore a possible cause for the observed elevated in-well flow velocities.

To assess the repeatability of the tracer-dilution test results, two separate tests were conducted at well 299-W22-50. A comparison of the tests indicates small, but discernable differences in the associated v_w estimates, i.e., Test #1 = 0.066 m/d; Test #2 = 0.046 m/d. Results for Test #2 are considered to be more representative based on the lower initial tracer concentration used (i.e., possible tracer concentration bias), and the longer tracer-dilution period exhibited.

A comparison of the observed depth/well velocity profiles provided information about permeability distribution within the well-screen sections at four of the wells. At wells 299-W10-24 (WMA TX-TY)and -W15-41 (WMA T) the highest flow velocities (and inferred permeabilities) were exhibited near the middle of the screen, with lowest flow velocities indicated near the top. Conversely, for well 299-W22-48 (WMA S-SX), the highest flow velocity was denoted near the top, with essentially little to no flow indicated for the lower part of the well screen. For well 299-W22-50 (southern boundary of WMA S-SX), relatively uniform depth/well velocity profiles were exhibited, indicating homogeneous conditions throughout the well-screen section. This condition was indicated for both tests conducted at the well site.

Table 4

Table 4 lists pertinent information pertaining to the tracer pumpback tests performed. As noted previously, several wells exhibited vertical flow conditions during the tracer-dilution tests (denoted by VF in the table). The fact that tracer only was emplaced into the aquifer within a small portion of the well screen, seriously impacts the assumptions of the test (which will be discussed in detail in the subsequent PNNL technical report). The tracer pumpback results for those wells affected by vertical flow conditions are highly questionable, and should not be used for quantitative assessment. The estimates calculated from the tests, however, are provided in the table (in parentheses) for only comparison/informational purposes.

Estimates for n_e for the reportable tests ranged between 0.068 and 0.257 (note: Test #2 for well 299-W22-50 is believed more representative, due to the fact that longer tracer drift times are less affected by well effects). This range for n_e falls within the common range usually reported for semi-consolidated to unconsolidated alluvial aquifers of 0.05 to 0.30, and brackets the large-scale values for specific yield, S_y ($S_y \approx n_e$) of 0.11 and 0.17, reported in Newcomb and Strand (1953) and

Wurstner et al. (1995), respectively for the 200-West Area. These large-scale analysis values were based on analyzing the growth and decline of the groundwater mound beneath the 200-West Area, that were associated with water disposal practices in the area.

Estimates for v_a for the reportable tests ranged between 0.013 and 0.374 m/d, and generally fall within a factor of 2 of the calculated in-well flow velocities, v_w. As noted previously for v_w at well 299-W15-41, the observed estimate for v_a of 0.374 m/d at this well site may be elevated due to affects imposed by operation of the adjacent 200-ZP-1 pump and treat system.

Table 5

Table 5 lists groundwater flow characterization results pertaining to determination of groundwater-flow direction and hydraulic gradient, I, conditions at the various test sites during the times of tracer testing. Groundwater-flow direction and hydraulic gradient were calculated using the commercially available WATER-VEL (In-Situ, Inc. 1991) software program. Water-level elevations from neighboring, representative wells were used as input with the WATER-VEL program to calculate groundwater-flow direction and hydraulic gradient conditions during the detailed characterization period. The program utilizes a linear, two-dimensional trend surface (least squares) to randomly located hydrologic head or water-level elevation input data. This method is similar also to the linear approximation technique described by Abriola and Pinder (1982) and Kelly and Bogardi (1989). A report that demonstrates the use of the WATER-VEL program for calculation of groundwater-flow velocity and direction is presented in Gilmore et al. (1992) and Spane (1999).

Calculations of I listed in Table 5 were used for estimates of n_e and v_a shown in Table 4. The indicated easterly groundwater flow directions for WMA S-SX and T sites and the southerly groundwater flow direction for the TX-TY directions is consistent with previous generalizations presented in Hartman et al. (1999) for these areas.

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Table 1. Detailed Hydrologic Characterization Elements

Characterization Element	Activities	Results
Groundwater Flow Characterization	Trend-surface analysis of well water-level data	Quantitative determination of groundwater flow direction and hydraulic gradient
Barometric Response Evaluation	Well water-level response characteristics to barometric changes	Aquifer/well model identification, vadose zone property characterization, correction of hydrologic test responses for barometric pressure fluctuations
Slug Testing	Multi-stress level tests conducted at each well site	Local K_h, T of aquifer surrounding well site.
Tracer-Dilution Testing	Monitoring dilution of administered tracer at injection well site	Vertical distribution of K_h, groundwater flow velocity at injection well location
In-Well Vertical Tracer Test	Monitoring the vertical movement of tracer within the well screen	Determination of vertical flow within the monitoring well screen section
Tracer Pumpback	Pumping/monitoring of recovered tracer and associated pressure response in monitoring wells	Large-scale, interwell n_e, K_h, K_v/K_h, K_{hx}/K_{hy}, T, S, S_y
Step-Drawdown Test	Determine well water-level response to selected pumping rates	Well loss characteristics

Hydrologic parameters:

K_h = horizontal hydraulic conductivity; (L/T)
K_v/K_h = vertical anisotropy; (dimensionless)
K_{hx}/K_{hy} = horizontal anisotropy; (dimensionless)
T = transmissivity; (L^2/T)
S = storativity; (dimensionless)
n_e = effective porosity; (dimensionless)

Table 2. FY-99 and FY-00 Hydraulic Property Test Analysis Summary for WMA TX-TY, T, and S-SX

WM	Well	Hydrologic Characterization Tests		
		Slug Test	Constant-Rate Pumping Test	
		Equivalent Hydraulic Conduct K_e m/d	Equivalent Hydraulic Conduct K_e m/d	Transmissivity T m^2/d
TX-TY	299-W10-26	1.40 – 1.95	1.5	82
	299-W14-13	1.66 – 2.43	2.4	135
	299-W14-14	2.44 – 2.87	--	--
	299-W15-40	0.88 – 1.22	--	--
	299-W15-41	15.1 – 19.5*	19.6**	1130**
T	299-W10-23	1.65 – 2.35	--	--
	299-W10-24	1.04 – 1.68	1.2	66
S-SX	2-W22-48	1.55 - 1.98*	1.81**	127**
	2-W22-49	6.92 – 8.20*	7.17**	520**
	2-W22-50	5.18 – 5.46*	5.24**	385**

Note: unless otherwise indicated, slug test analysis range represents the average analysis value for the Bouwer and Rice and type-curve methods

* slug test results do not include analysis results for Bouwer and Rice method; listed range will be updated when analysis results are complete in FY-01

** preliminary pumping test analysis values, subject to revision; to be documented in FY-01

K_e assumes aquifer with uniform hydraulic conductivity value

-- constant-rate pumping test not conducted at the well site

Table 3. FY-99 and FY-00 Tracer-Dilution Test Analysis Summary for WMA TX-TY, T, and S-SX

WMA	Well	Test Interval m, btoc[a]	Tracer-Dilution Test Results					
			Date Test Initiated	Total Dilution Time t_d min	Average Initial Tracer Concent. C_o mg/L	Average Final Tracer Concent. C_t mg/L	Average Well Flow Velocity n_w m/d	Range Well Flow Velocity n_{wz} m/d
TX-TY	2-W10-26	67.4 - 77.8	4/23/99	7,259	219	<1.0	vf (0.086)	vf (downward)
	2-W14-13	67.1 - 77.9	3/26/99	8,575	VF	VF	VF	VF (downward)
	2-W15-41	66.3 - 71.1	5/8/00	2,714	152	< 1.5	0.311	0.232 – 0.401*
T	2-W10-24	72.4 - 82.6	4/9/99	17,455	148	26	0.012	0.009 – 0.017*
S-SX	2-W22-48	70.5 – 74.3	5/11/00	15,730	141	39	0.007*	0.002 - 0.023**
	2-W22-49	67.3 – 71.9	4/17/00	4,175	145	4.0	vf (0.086)	vf (upward)
	2-W22-50	67.5 – 71.9	5/1/00 (Test #1)	5,765	190	5.2	0.066	relatively uniform
			5/26/00 (Test #2)	7,240	148	6.5	0.046	relatively uniform

* permeability profile indicates highest permeability (highest flow velocity) near the middle of well screen; lowest permeability near top

** permeability profile indicates highest permeability (flow velocity) near top of well screen, becoming progressively lower with depth within well screen

C_o estimated initial tracer concentration based linear back-projection of average well screen conditions

C_t average observed well-screen tracer concentration at termination of test

ν_w average groundwater flow velocity within well

ν_{wz} groundwater flow velocity range within well determined from individual probe/depth-settings

vf slight vertical flow conditions detected adversely affect tracer test results; vertical flow direction indicated in parentheses

VF significant vertical flow conditions in well invalidating tracer-dilution test; vertical flow direction indicated in parentheses

Table 4. FY-99 and FY-00 Tracer-Pumpback Test Analysis Summary for WMA TX-TY, T, and S-SX

			Hydrologic Characterization Tests						
						Tracer Pumpback Test			
WMA	**Well**	**Aquifer Thickness b m**	**Pumping Rate Q L/min**	**Hydraulic Gradient m/m**	**Trans-missivity T m^2/d**	**Tracer Drift Time t_d min**	**Tracer Recovery Time t_p min**	**Effective Porosity, n_e**	**Ground-Water Flow Velocity n_a m/d**
TX-TY	2-W10-26	55.0	39.5	0.00073	82	7,259	16.0	vf (0.010)	vf (0.124)
	2-W14-13	55.0	48.9	0.00073	135	8,575	43.3	VF (0.009)	VF (0.191)
	2-W15-41	57.6	60.4	0.00129	1130*	2,714	109.0	0.068*	0.374*
T	2-W10-24	54.0	41.2	0.00172	66	17,455	37.1	0.072	0.029
S-SX	2-W22-48	70.1	7.0	0.00180	127*	15,730	159.1	0.257*	0.013*
	2-W22-49	72.5	42.2	0.00206	520*	4,175	14.9	VF (0.671*)	VF (0.022)
	2-W22-50	73.5	28.5 (Test #1)	0.00206	385*	5,765	43.4	0.354	0.030
			29.2 (Test #2)	0.00206	385*	7,240	108.8	0.221	0.049

* preliminary hydraulic property estimate values (T); tracer pumpback results subject to revision
t_d time tracer allowed to drift from well into surrounding aquifer prior to pumpback
t_p time required to recover 50% of the tracer mass during the pumpback
ν_a groundwater flow velocity within aquifer
(vf) slight vertical flow conditions in well detected; tracer test estimates for n_e and v_a are questionable
VF significant vertical flow conditions in well detected; tracer test estimates for n_e and v_a are highly questionable

Table 5. FY-99 and FY-00 Groundwater Flow Characterization Results Based on Trend-Surface Analysis for WMA TX-TY, T, and S-SX

WMA	Well	Measurement Date	Trend-Surface Analysis Results		
			Groundwater Flow Direction 0° = East; 90° = North	Hydraulic Gradient m/m	Wells Used in Analysis
TX-TY	2-W10-26	5/3/99	288°	0.00073	299-W10-17, -W10-18, -W14-12, -W15-12, -W15-22
	2-W14-13	5/3/99	288°	0.00073	299-W10-17, -W10-18, -W14-12, -W15-12, -W15-22
	2-W15-41	5/8-11/00	286°	0.00129	299-W14-5, -W14-6, -W14-14, -W15-40, -W15-41
T	2-W10-24	4/21/99	5°	0.00172	299-W10-8, -W10-12, -W10-22, -W10-24, -W11-23, -W11-27
S-SX	2-W22-48	5/18/00	2°	0.00180	299-W22-45, -W22-48, -W23-13
	2-W22-49	5/31/00	1°	0.00206	299-W22-49, -W22-50, -W23-14, -W23-15
	2-W22-50	5/31/00	1°	0.00206	299-W22-49, -W22-50, -W23-14, -W23-15

Appendix B

Hydraulic Gradients and Groundwater Flow Direction Determinations

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Internal Distribution

Date	September 26, 2000	D.B. Barnett M.J. Hartman F.N. Hodges
To	V.G. Johnson	S.P. Luttrell J.P. McDonald
From	F.A. Spane	S.M. Narbutovskih D.R. Newcomer
Subject	Groundwater Flow Characterization Within the WMA S-SX	M.D. Sweeney W.D. Webber File/LB

This letter report presents the results of a detailed groundwater flow characterization (i.e., groundwater flow direction and hydraulic gradient determination) within and proximate to the Waste Management Area (WMA) S-SX, in the Hanford Site 200 West Area. Most of the analytical discussion contained in this letter report was taken primarily from Spane (1999). The analytical methods were applied to representative well measurements available within the WMA S-SX.

Introduction

Groundwater flow characterization is important as it pertains to predicting and monitoring groundwater contaminant migration within the Hanford Site. Accurate delineation of local groundwater-flow direction and hydraulic gradient conditions within study areas of small size and/or having low gradient conditions, however, can be difficult. A method that facilitates groundwater flow characterization in such areas is the use of trend-surface analysis of representative monitoring well water-level measurements (see Spane 1999).

Various factors can affect the accuracy of well water-level measurements and how they are used to determine hydraulic head and to infer groundwater-flow conditions within an aquifer. These factors include measurement error, well fluid-column density conditions, and external stress effects. Measurement error includes the cumulative effect of instrument and measuring point elevation errors, borehole deviation, and random measurement factors, such as operator error. Systematic components of measurement error can be evaluated qualitatively by assessing the relative influence of individual well water-level measurements on the calculated groundwater flow characteristics. This was done using sensitivity analysis (i.e., "jack-knife" analysis), wherein each well's measurement was removed individually from the selected well data set, and then subjected to trend-surface analysis. Results from this sensitivity analysis suggest that systematic measurement errors were not significant for studying groundwater flow characteristics in the WMA S-SX.

Well fluid-column density conditions relate to factors that affect the height of a fluid column in a well above a known elevation datum. Factors that can affect fluid-column density include fluid temperature, salinity, pressure, dissolved gas content, multiphase conditions, and gravitational acceleration effects. Generally, these factors are only significant for deep or thick aquifers having long fluid-column lengths, which was not the case for this investigation.

Natural external stresses that can influence well water-level measurements include barometric effects, tidal or river-stage fluctuations, and earth tides. Earlier papers have addressed these effects on well water-level measurements within confined and unconfined aquifer systems (e.g., Jacob 1940; Ferris 1963; Bredehoeft 1967; Weeks 1979; Hsieh et al. 1988; Erskine 1991). Only recently, however, has the importance of removing external stress factor effects from water-level measurements for wells monitoring shallow unconfined aquifer systems been recognized (see Rasmussen and Crawford 1997, and Spane 1999). This letter report focuses on:

- evaluating barometric effects on WMA S-SX well water-level measurements
- determining existing groundwater flow characteristics (flow direction, gradient)
- assessing any changes in groundwater flow characteristics with time, and
- provides recommendations for improving monitoring of groundwater flow characteristics in this area

Results from this letter report will be useful not only for groundwater flow characterization, but also for the design and placement of future monitoring wells in this area of the Hanford Site.

Data Discussion

The 200-West Area has witnessed significant changes in the water table due to wastewater disposal activities in the area. Of particular importance to the study area were wastewater disposals to U Pond complex (located approximately 1000 m west of the WMA S -SX, which received approximately 60% of the total wastewater released in the 200-West Area (Newcomer 1990). These wastewater disposal activities caused discernable changes in the prevailing groundwater flow pattern and formation of a large groundwater mound with elevated water-table conditions approximately 20 m over pre-disposal conditions (Hartman and Dresel 1998).

With the decommissioning of U Pond in 1984, a significant decrease in wastewater disposal and associated decline in water-table elevation were exhibited across the 200-West Area. For example, Hartman and Dresel (1998) report a 6 m decline between 1984 and 1997. The decline in the water table and changes in groundwater flow characteristics are expected to continue with future decreases in wastewater releases to 200-West Area disposal facilities.

To evaluate existing and temporal groundwater flow characteristics within the study area, well water-level measurements were evaluated from RCRA monitoring wells within the WMA S-SX. Figure 1 shows the locations of monitoring wells having data for groundwater flow characterization. Table 1 lists pertinent information concerning well completion and current monitoring conditions for the RCRA wells.

Groundwater Flow Characterization

The effects of barometric pressure fluctuations on well measurements used in groundwater flow characterization have been discussed in detail in Rasmussen and Crawford (1997) and Spane (1999). Based on these reports, it is apparent that atmospheric pressure fluctuations may cause temporal variations/changes in groundwater-flow patterns (flow velocity, flow direction) within unconfined aquifers that exhibit low hydraulic gradient conditions and variable vadose zone properties (e.g., 200 West Area of the Hanford Site). This is due to the areal variation in transmission of atmospheric pressure to the water-table surface, which is part of the total hydraulic head governing groundwater flow.

As noted in Spane (1999), for the determination of groundwater-flow direction and hydraulic gradient, total head (**not** well water-level elevation) should be the hydrologic parameter analyzed. For wells not exhibiting significant wellbore-storage/well-skin effects, total head can be calculated for confined and unconfined aquifers by adding the incremental barometric head (as compared to the reference barometric value) directly to the well water-level (elevation) measurement. Groundwater-flow direction and hydraulic gradient can be determined by standard trend-surface-fitting methods (or three-point problems) using total head measurements obtained from monitoring wells that meet the following criteria listed in Spane (1999):

- are along the same hydrologic flow plane (i.e., planar potential surface)
- are measured close in time (e.g., within 12 h [1 to 4 h for low-gradient areas])
- monitor similar depth intervals within the respective hydrogeologic unit
- display similar dynamic well-response characteristics (e.g., to barometric fluctuations)
- are not significantly affected by well-skin effects.

To evaluate the sensitivity of groundwater-flow direction and hydraulic gradient determinations within the WMA S-SX, standard frequency (e.g., quarter-annual) Hanford Site water-level data were analyzed for RCRA wells used to monitor conditions surrounding this facility (see Figure 1). This site was identified in Hartman and Dresel (1998) as being an intermediate-hydraulic gradient area (~ 0.002), and having a predominant, southeasterly, groundwater-flow direction over the period 1992-1997 (Johnson and Chou, 1998).

Barometric Response Analysis

To examine the temporal effects of barometric pressure fluctuations on monitoring well water-level measurements, three S-SX RCRA wells were selected for detailed monitoring, i.e., hourly measurements over 7 to 10 day periods. The wells selected for detailed monitoring included two newly constructed wells (299-W22-48 and 299-W22-49), and a previously established well 299-W23-15. Locations for the selected well are shown in Figure 1, with pertinent well construction details provided in Table 1.

Figure 2 shows the effect of barometric pressure on well water-level measurements at well 2-W22-49. As indicated, well water levels exhibit an inverse relationship to barometric pressure variations. During the 9-day monitoring period in 2000, barometric pressure varied by 0.26 m, while water levels within the well varied by 0.13 m. Results from detailed multiple-regression analysis, as described in Rasmussen and Crawford (1997) and Spane (1999), indicate a delayed time-lag response pattern (with a best-fit, time-lag dependence of 38-hr) exhibited for all three sites. Figure 3 and Table 2 show a comparison of the barometric time-lag characteristics for all 3 sites over the 38-hr, time-lag period. The barometric response pattern shapes exhibited in Figure 3 for the wells are quite similar and are characteristic of a delayed, unconfined aquifer model. For comparison purposes, the barometric response patterns for predicted unconfined aquifer behavior were also calculated for the existing water-table depth conditions, using the Weeks (1979) vadose model (WBAR). As shown, the unconfined aquifer behavior bounds the later time-lag response behavior (i.e., for time lags > 20 hr) for a vadose zone having pneumatic diffusivities ranging between 0.02 and 0.03 m^2/s. It should be noted that the predicted time-lag response does not match the observed early time-lag response characteristics (i.e., < 10 hr) at the wells. This is attributed to the fact that the vadose zone model used, does not account for either wellbore storage/"skin effects" (i.e., well/formation inefficiencies or damage) or the boundary condition where the water table occurs within the well screen (allowing direct atmospheric pressure propagation to occur from the well to the water table from the well in addition to vertically through the vadose zone). The impact of both factors would be more prevalent for early time-lags and would act to subdue the well barometric response characteristics for early-time lag periods. This composite effect is what is believed controlling the barometric early-time response pattern exhibited at all three sites.

The overall similarity in barometric response characteristics also suggests that vadose zone conditions are relatively uniform over the WMA S-SX. Well water-level measurements, therefore, taken fairly close in time for the S-SX RCRA monitoring well network are not likely to be adversely affected by temporal changes in barometric pressure, when used collectively for detailed groundwater flow characterization. For situations where measurements are taken over extended periods of time, adjustments to well hydraulic head measurements can be made by adding the head amount equal to the observed barometric pressure at the time of measurement minus to long-term reference barometric pressure for the site. Spane (1999) recommends that a long-term reference barometric pressure of 10.087 m be used for Hanford Site groundwater characterization studies.

Figure 4 shows the match of predicted well water-level response at well 299-W22-49 for the observed barometric pressure record (shown in Figure 2) and multiple-regression coefficients for a 38-hr time-lag analysis (listed in Table 2). As shown in the figure, the predicted response provides a close match for the observed well water levels, and provides an effective means for removing barometric pressure fluctuations from the well water-level record. As noted in Spane (1999), the ability to remove barometric pressure effects from well water-level measurements is particularly important for quantitative analysis of long duration hydrologic tests, e.g., pumping tests. Similar barometric analysis matching/correction results were observed for wells 299-W22-48 and 299-W23-15, but are not included in this report.

Trend-Surface Analysis Results

Available RCRA monitoring well data were quantitatively evaluated for groundwater-flow characterization using some of the screening criteria listed previously. Because detailed barometric response analysis data were not available for all RCRA wells, a general evaluation of the temporal water-level-response characteristics for wells completed at similar depth intervals was performed for data collected during calendar years 1992 through 1999. Figure 5 shows the similarity in dynamic well-response characteristics exhibited for the seven monitoring wells selected for detailed groundwater-flow characterization. The overall declining water-level elevation trend pattern is consistent with the general decrease in total wastewater disposal within the 200-West Area during the mid-1980's as previously discussed.

To facilitate quantitative determination of groundwater-flow direction and hydraulic gradient conditions, the commercially available WATER-VEL (In-Situ, Inc. 1991) software program was utilized. Water-level elevation and calculated total head values were used with the WATER-VEL program to calculate groundwater-flow direction and hydraulic gradient conditions over the measurement period. The program utilizes a linear, two-dimensional trend surface (least squares) to randomly located hydrologic head or water-level elevation input data. This technique is accurate as long as the two-dimensional linear approximation is applicable (i.e., no significant vertical groundwater-flow gradients exist within the aquifer). This method is similar also to the linear approximation technique described by Abriola and Pinder (1982) and Kelly and Bogardi (1989). A report that demonstrates the use of the WATER-VEL program for calculation of groundwater-flow velocity and direction is presented in Gilmore et al. (1992) and Spane (1999).

Because surrounding well water-level measurements were collected, in some cases, over a period of several days, the effects of barometric pressure fluctuations may be expected to exert a discernible influence in calculating groundwater-flow direction and hydraulic gradient. Figure 6 shows the relationship of well water-level elevation measured at the 7 different S-SX RCRA monitoring wells and the barometric pressure fluctuation pattern over the field measurement period (August 6 - 12, 1998). As shown, the barometric pressure varied by 0.11 m during the actual period of well measurements, which compares with a maximum 0.50-m water-level elevation difference between wells. Although the barometric pressure variation comprises only a small percentage (i.e., ~20%) of the observed S-SX monitoring well water-level elevation measurements, at other Hanford low hydraulic gradient sites (e.g., 200 East Area) barometric pressure fluctuations may actually exceed observed areal well water-level elevation differences. To minimize the effects of barometric pressure fluctuations within low-gradient areas, Spane (1999) notes that all well water-levels should be measured over a short-period of time (i.e., within 1 to 4 hrs), with more emphasis placed on measurements obtained during summer months, when diurnal barometric pressure fluctuations are relatively small.

To quantitatively assess the groundwater-flow characteristics within the WMA S-SX over time, both observed well water-level elevation measurements and calculated total head values (based on the recommended Hanford Site barometric reference value of 10.087 m) were analyzed. Table 3 lists the results of quantitative trend-surface analysis for seven summer-month, measurement periods for the seven existing RCRA monitoring wells over the 1993 to 1999 time period. As shown, nearly identical results were obtained using either water-level elevation or total head measurements. This close correspondence between measurement results is expected, due to the relatively low impact of diurnal barometric pressure fluctuations on well water-level elevation measurements during summer months in the WMA S-SX.

As indicated in Table 3, the results of the trend-surface analysis provide a consistent pattern of a progressively increasing eastward groundwater flow direction (307° to 350°) over the seven-year time period. The hydraulic gradient exhibited a less consistent pattern (ranging between 2.6E-03 to 1.6E-03), but generally declined overall during the analysis period. The progressively easterly groundwater flow and decreasing hydraulic gradient pattern is consistent with cessation of wastewater disposal activities to U Pond in the 200-West as previously discussed. The average groundwater flow direction (328°) and hydraulic gradient (2.05E-03) are consistent with previously reported conditions in Johnson and Chou (1998) for the WMA S-SX.

Because of the added importance in understanding contaminant groundwater transport conditions within the southern SX area, the three RCRA monitoring wells in this immediate area (wells 299-W22-39, 299-W22-46, 299-W23-15) were selected for additional analysis. Table 4 lists the results of the trend-surface analysis for the same measurement time period. As shown, similar groundwater flow characteristics are exhibited overall, although a more consistent easterly groundwater flow direction is indicated. An average groundwater flow direction of 349° and hydraulic gradient of 2.17E-03 are indicated for the southern SX area for the 1993 – 1999 measurement time period.

To examine any apparent groundwater flow pattern differences across the WMA facility, a detailed evaluation for the northern S area was also initiated. Three RCRA monitoring wells in this immediate area (wells 299-W22-44, 299-W22-45, 299-W23-13) were selected for this areal analysis. Table 5 lists the results of the trend-surface analysis for the same measurement time period. As shown, similar temporal groundwater flow characteristics were exhibited for the northern S area as were exhibited for the entire WMA S-SX (Table 3). Results from the trend-surface analysis provide a consistent pattern of a progressively increasing eastward groundwater flow direction (293° to 354°) over the seven-year time period within the northern S area. An average groundwater flow direction of 324° and hydraulic gradient of 1.99E-03 are indicated for this area for the 1993 – 1999 measurement period.

Recommendations

Based on the results of this study, the following recommendations are provided for improving the characterization of groundwater flow conditions within the WMA S-SX:

- trend-surface analysis methods should be used for delineating groundwater flow direction/gradient conditions
- more reliance on trend-surface analysis results should be given for measurements obtained during summer months, when diurnal barometric fluctuations are low

- water levels within the S-SX RCRA monitoring wells should be measured on the same day, and preferably within a 4-hr period to minimize the impact of barometric fluctuation effects
- replacements for upgradient monitoring wells 299-W23-13 and 299-W23-14 will be necessary soon, due to continued decline in areal water levels
- efforts should be initiated to characterize and remove the effects of measurement error (e.g., borehole deviation) from well water-level measurements used to characterize groundwater flow conditions within the WMA S-SX

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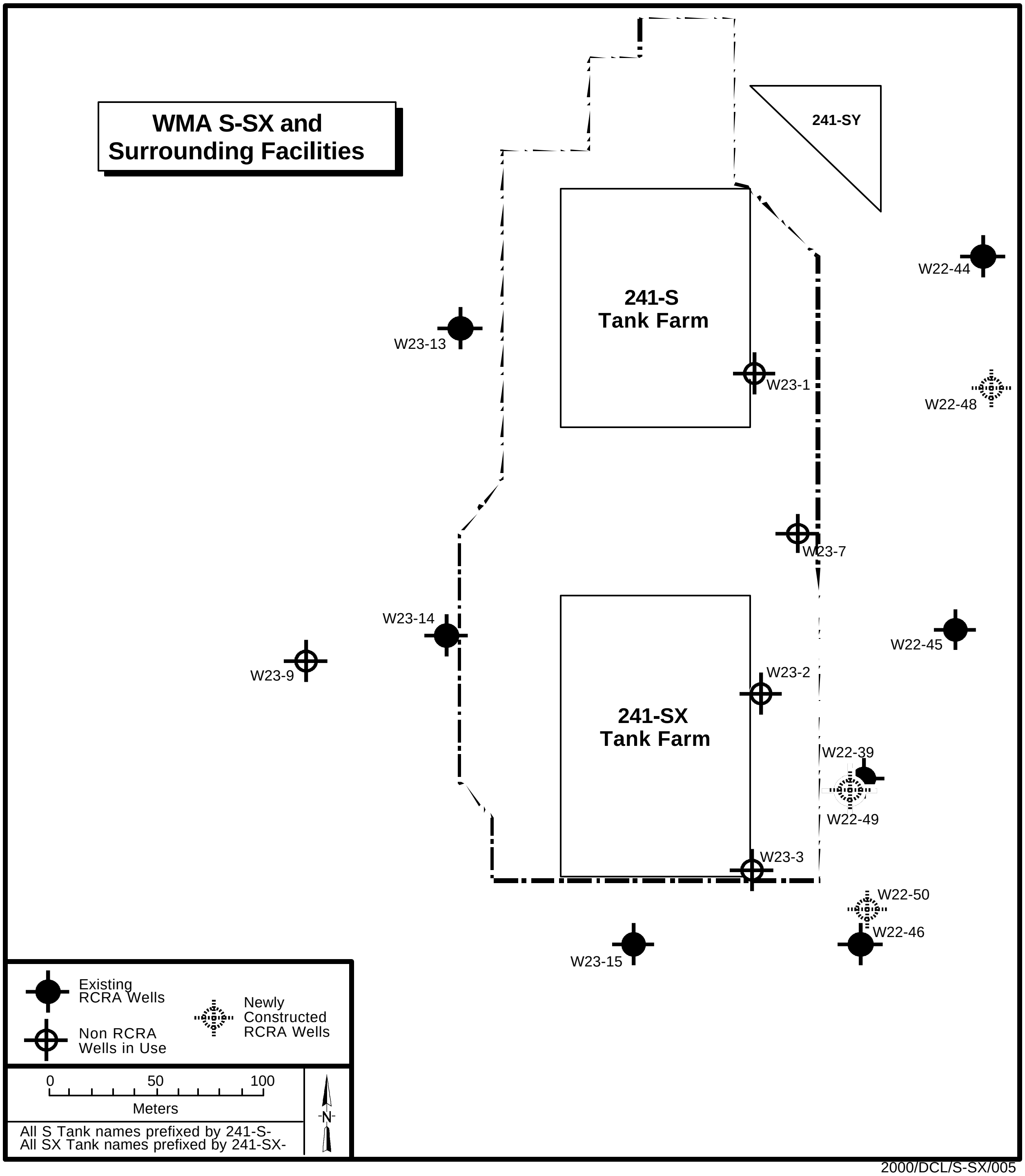

Figure 1. Location Map of Wells Monitoring the WMA S-SX, 200-West Area.

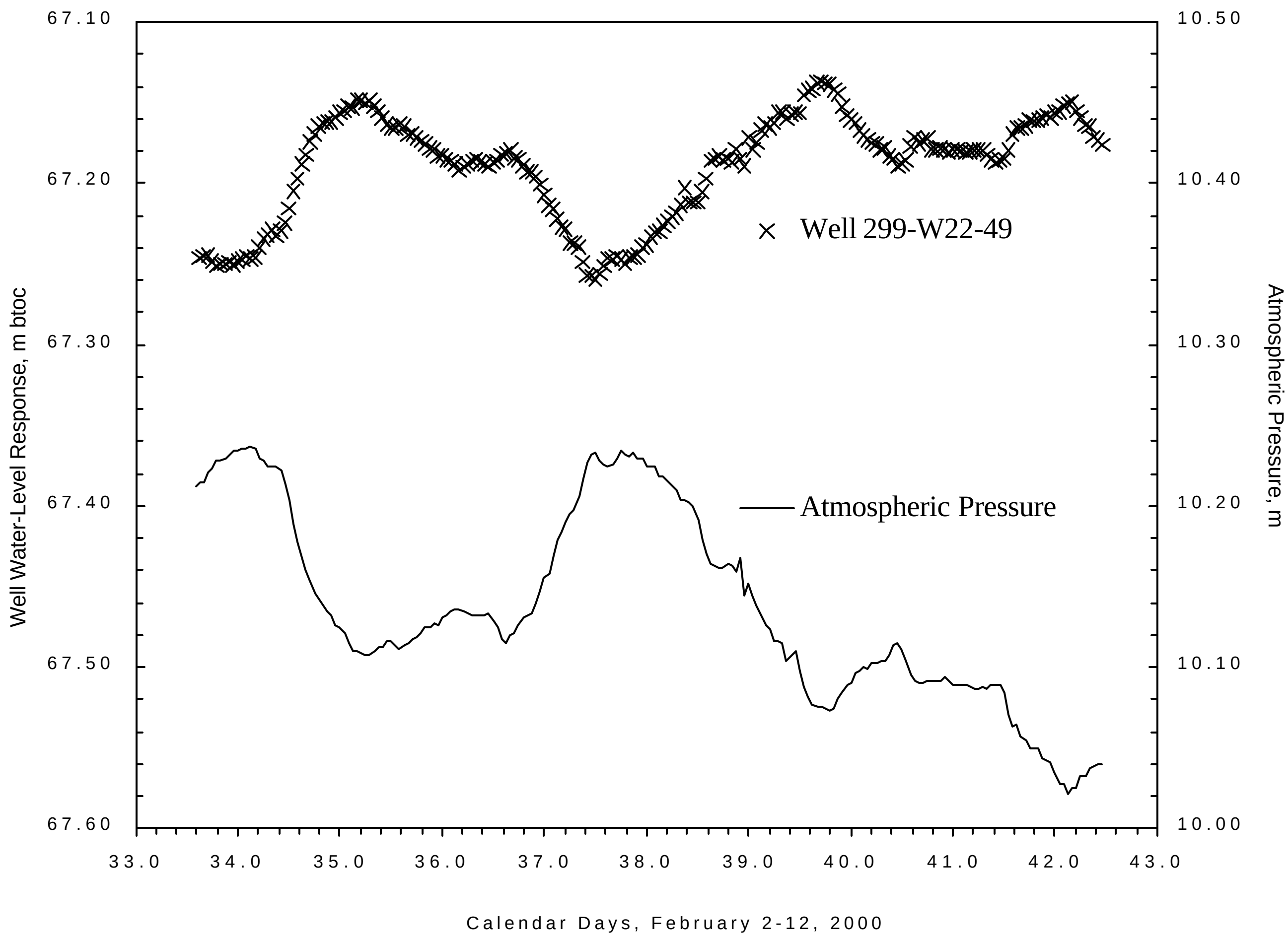

Figure 2. Well Water-Level and Barometric Pressure Measurements for Well 299-W22-49.

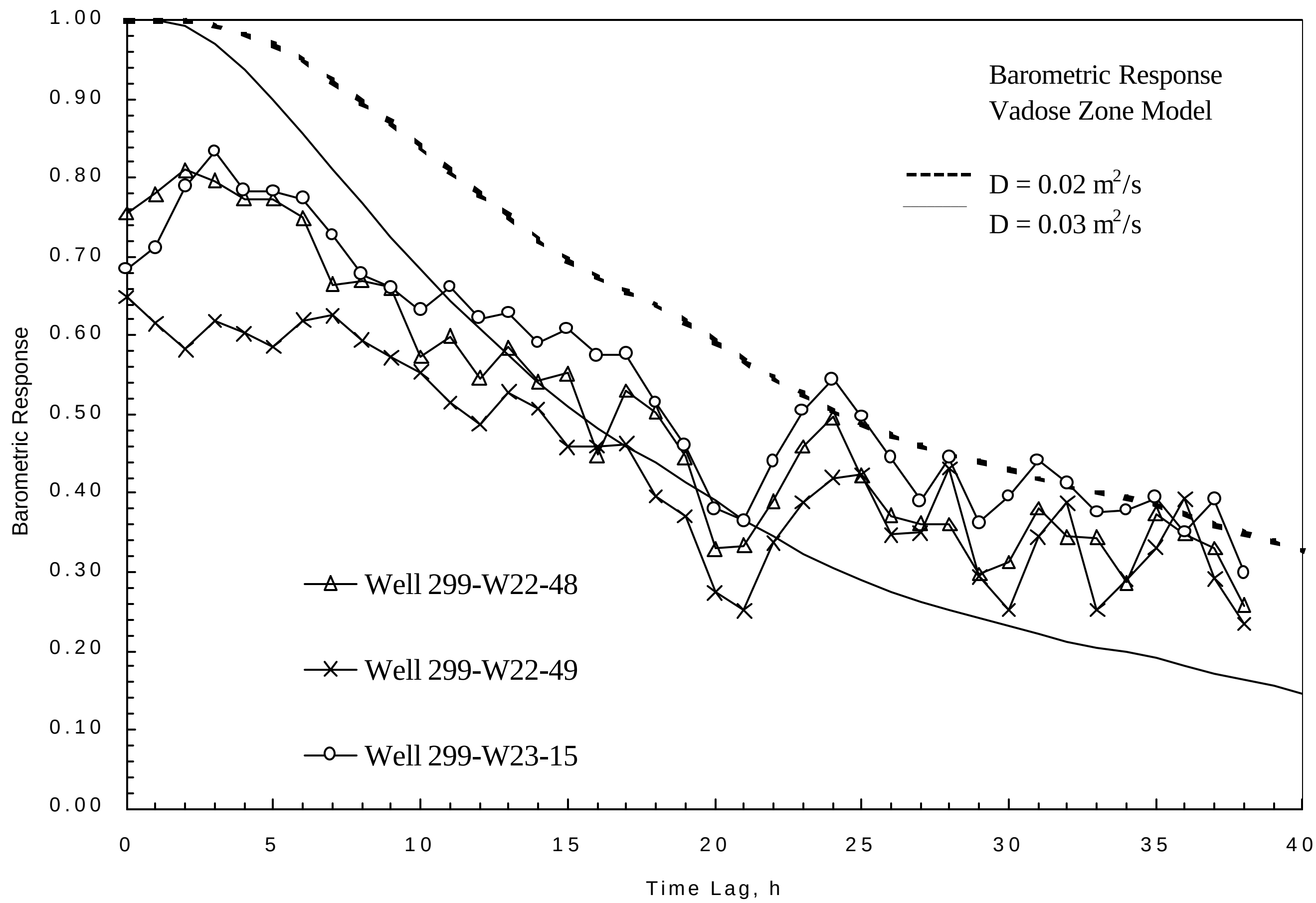

Figure 3. Water-Level Barometric Response Patterns for Wells 299-W22-48, 299-W22-49, and 299-W23-15.

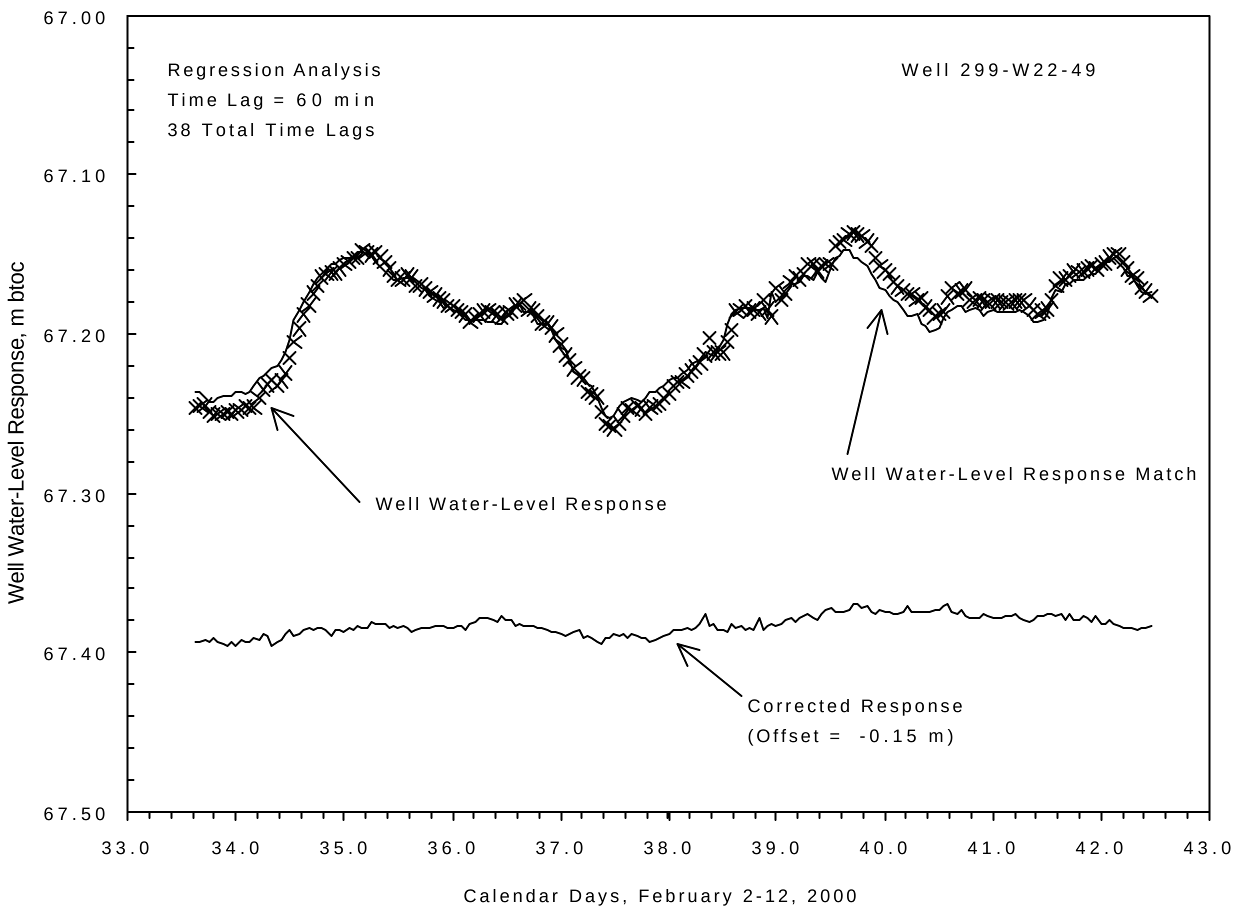

Figure 4. Multiple-Regression Match and Barometric Correction of Well Water-Level Response for Well 299-W22-49.

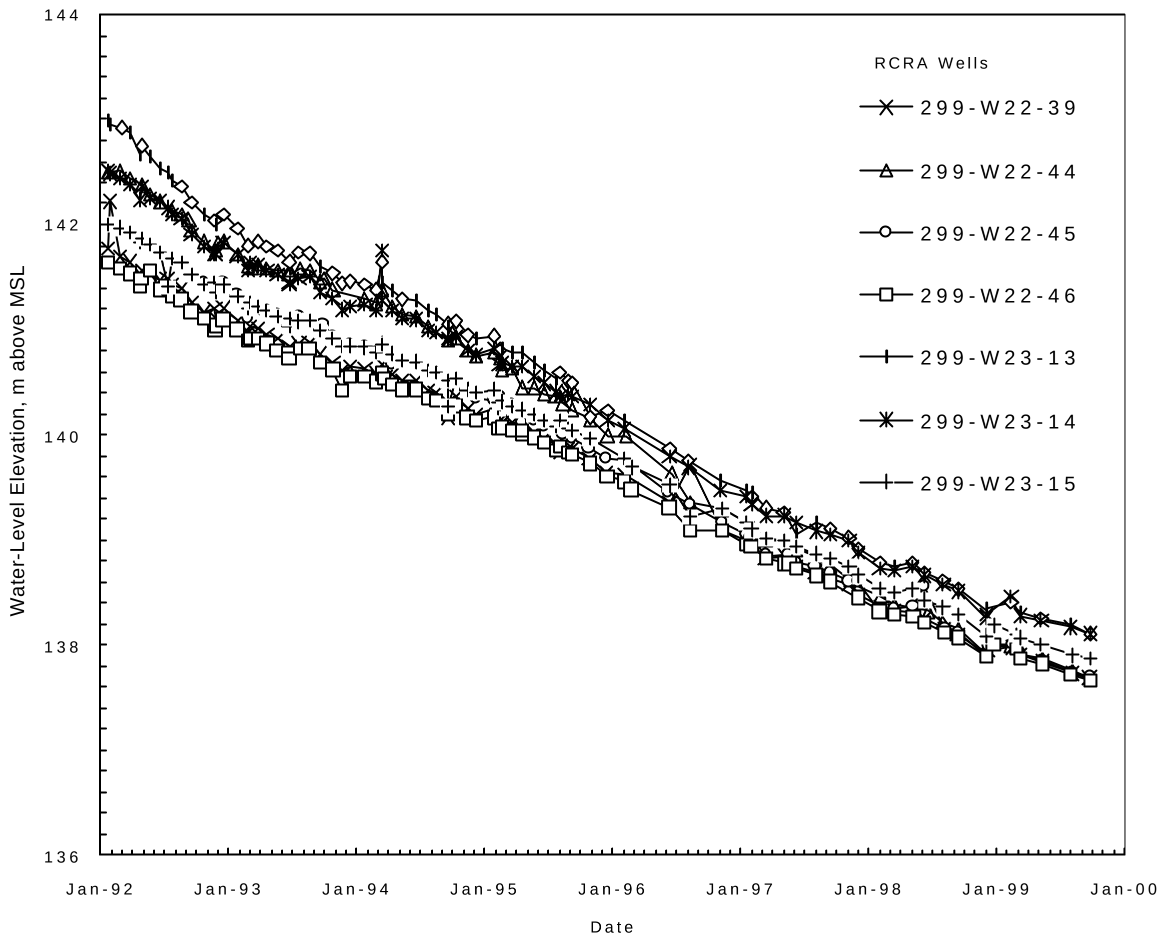

Figure 5. Comparison of Well Water-Level Elevation Response for Selected RCRA Wells Monitoring the WMA S-SX

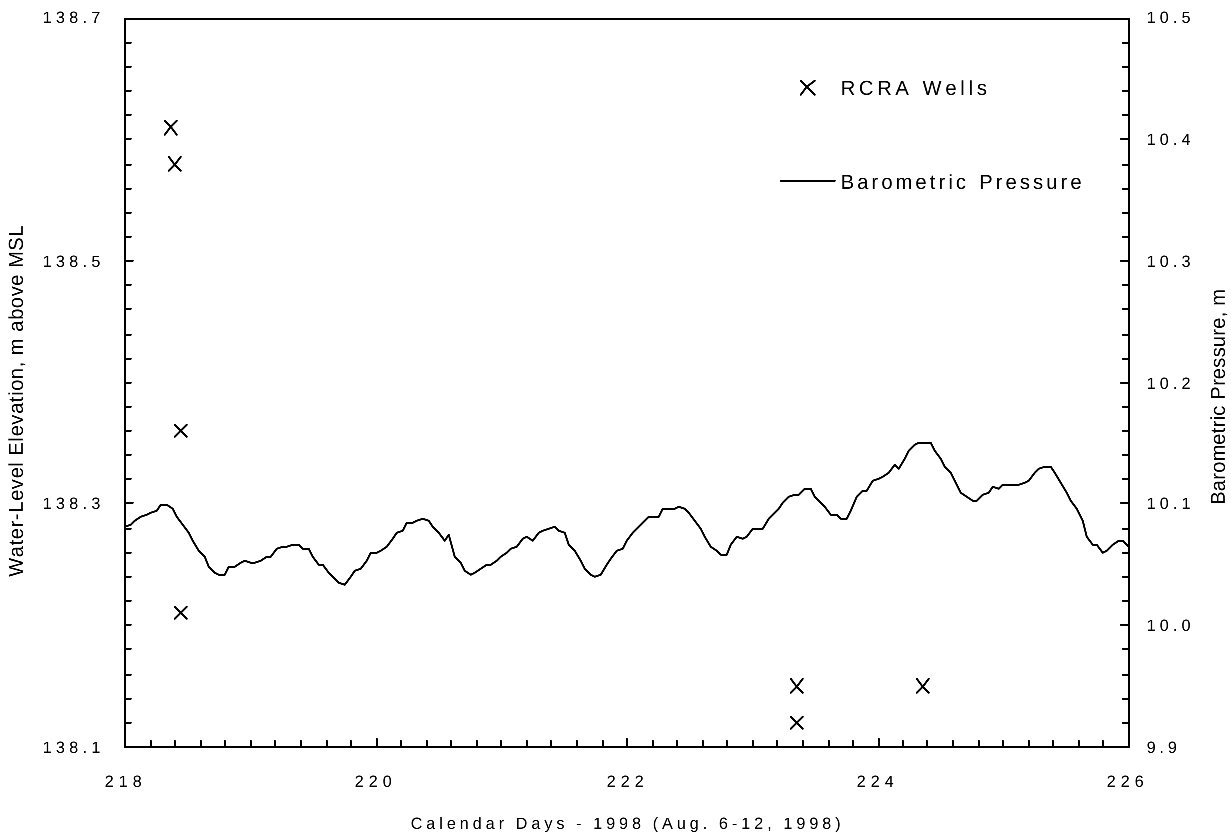

Figure 6. Comparison of Well Water-Level Elevation and Atmospheric Pressure for Selected S-SX RCRA Monitoring Wells, August 6 – 12, 1998.

Table 1. Pertinent Well Completion Information for RCRA Wells Monitoring the WMA S-SX Facilities

Well	Completion Date M/Yr	Well Screen Length m, bgs	Water-Level Depth, m, bgs	Water-Column Length Above Well Screen Bottom, m
299-W22-39	2/91	60.90 – 67.45	67.06 (9/99)	0.39
299-W22-44	11/91	62.51 – 73.82	70.07 (9/99)	3.75
299-W22-45	9/92	60.38 – 71.29	65.64 (1/00)	5.65
299-W22-46	11/91	58.80 – 69.77	67.98 (9/98)	1.79
299-W22-48	11/99	68.96 – 73.53	69.55 (1/00)	3.98
299-W22-49	11/99	66.42 – 70.99	66.34 (1/00)	4.65
299-W22-50	1/00	66.43 –71.00	66.83 (1/00)	4.17
299-W23-13	12/90	59.71 – 66.20	66.05 (9/99)	0.15
299-W23-14	4/91	59.13 – 65.62	65.35 (9/99)	0.27
299-W23-15	12/91	56.60 – 67.79	62.06 (1/00)	5.73

Table 2. Multiple-Regression Analysis for Wells 299-W22-48, 299-W22-49, and 299-W23-15.

	Well Water Level / Barometric Regression Analysis					
	299-W22-48		299-W22-49		299-W23-15	
Time Lag Hr	Regression Coefficient	Regression Coefficient Sum[a]	Regression Coefficient	Regression Coefficient Sum[a]	Regression Coefficient	Regression Coefficient Sum[a]
0	-0.7558	0.7558	-0.6492	0.6492	-0.6843	0.6843
1	-.00240	0.7798	0.0339	0.6153	-0.0278	0.7121
2	-.00309	0.8107	0.0326	0.5827	-0.0773	0.7894
3	0.0139	0.7968	-0.0359	0.6186	-0.0443	0.8337
4	0.0229	0.7739	0.0159	0.6027	0.0492	0.7845
5	0.0006	0.7733	0.0169	0.5858	0.0012	0.7833
6	0.0233	0.7500	-0.0339	0.6197	0.0095	0.7738
7	0.0843	0.6657	-0.0062	0.6259	0.0464	0.7274
8	-0.0039	0.6696	0.0319	0.5940	0.0499	0.6775
9	0.0091	0.6605	0.0215	0.5725	0.0163	0.6612
10	0.0874	0.5731	0.0186	0.5539	0.0285	0.6327
11	-0.0261	0.5992	0.0388	0.5151	-0.0293	0.6620
12	0.0529	0.5463	0.0273	0.4878	0.0400	0.6220
13	-0.0390	0.5853	-0.0404	0.5282	-0.0065	0.6285
14	0.0429	0.5424	0.0208	0.5074	0.0373	0.5912
15	-0.0101	0.5525	0.0488	0.4586	-0.0174	0.6086
16	0.1042	0.4483	-0.0007	0.4593	0.0336	0.5750
17	-0.0819	0.5302	-0.0029	0.4622	-0.0015	0.5765
18	0.0270	0.5032	0.0658	0.3964	0.0615	0.5150
19	0.0568	0.4464	0.0254	0.3710	0.0546	0.4604
20	0.1170	0.3294	0.0967	0.2743	0.0804	0.3800
21	-0.0039	0.3333	0.0235	0.2508	0.0150	0.3650
22	-0.0571	0.3904	-0.0868	0.3376	-0.0759	0.4409
23	-0.0692	0.4596	-0.0513	0.3889	-0.0641	0.5050
24	-0.0367	0.4963	-0.0306	0.4195	-0.0398	0.5448
25	0.0743	0.4220	-0.0043	0.4238	0.0473	0.4975
26	0.0507	0.3713	0.0766	0.3472	0.0522	0.4453
27	0.0102	0.3611	-0.0027	0.3499	0.0551	0.3902
28	0.0009	0.3602	-0.0813	0.4312	-0.0546	0.4448
29	0.0629	0.2973	0.1376	0.2936	0.0824	0.3624
30	-0.0149	0.3122	0.0416	0.2520	-0.0342	0.3966
31	-0.0685	0.3807	-0.0929	0.3449	-0.0454	0.4420
32	0.0367	0.3440	-0.0430	0.3879	0.0284	0.4136
33	0.0006	0.3434	0.1362	0.2517	0.0371	0.3765
34	0.0571	0.2863	-0.0394	0.2911	-0.0019	0.3784
35	-0.0874	0.3737	-0.0400	0.3311	-0.0166	0.3950
36	0.0258	0.3479	-0.0616	0.3927	0.0445	0.3505
37	0.0176	0.3303	0.1012	0.2915	-0.0414	0.3919
38	0.0726	0.2577	0.0573	0.2342	0.0927	0.2992

a) Absolute values for regression coefficient summation

Table 3. Trend-Surface Hydraulic Gradient and Groundwater-Flow Direction Determinations for RCRA Wells Monitoring the WMA S-SX[(a)]

Date	Flow Direction, (0° = E; 90° = N)		Hydraulic Gradient (m/m)		Maximum Observed Difference, m: Water-Level Elevation/Total Head/Barometric Head
	Water-Level Elevation	Total Head	Water-Level Elevation	Total Head	
August 19-20, 1993	307°	307°	2.63E-03	2.64E-03	0.91 / 0.91 / 0.01
July 26, 1994	308°	308°	2.39E-03	2.39E-03	0.83 / 0.83 / 0.00
July 25, 27, 1995	313°	316	2.01E-03	2.04E-03	0.69 / 0.69 / 0.04
June 13-14, 20, 1996	325°	330°	1.69E-03	1.76E-03	0.55 / 0.55/ 0.05
August 6-7, 1997	337°	337°	1.65E-03	1.64E-03	0.50 / 0.50 / 0.03
August 6, 11-12, 1998	346°	346°	1.89E-03	1.81E-03	0.49 / 0.45 / 0.06
August 5, 9-11, 1999	347°	350°	1.91E-03	2.08E-03	0.48 / 0.50 / 0.07
Average Values (Standard Deviation)	326° (± 17.4°)	328° (± 17.7°)	2.02E-03 (± 3.61E-04)	2.05E-03 (± 3.59E-04)	0.64 / 0.63 / 0.04

(a) RCRA monitoring well network: 299-W22-39, -W22-44, -W22-45, -W22-46, -W23-13, -W22-14, -W22-15.

Table 4. Trend-Surface Hydraulic Gradient and Groundwater-Flow Direction Determinations for Southern RCRA Wells Monitoring the WMA SX Facility[(b)]

Date	Flow Direction, (0° = E; 90° = N)		Hydraulic Gradient (m/m)		Maximum Observed Difference, m: Water-Level Elevation/Total Head/Barometric Head
	Water-Level Elevation	Total Head	Water-Level Elevation	Total Head	
August 19, 1993	344°	347°	2.56E-03	2.53E-03	0.27 / 0.27 / 0.00
July 26, 1994	341°	341°	2.51E-03	2.51E-03	0.26 / 0.26 / 0.00
July 27, 1995	344°	344°	2.18E-03	2.18E-03	0.23 / 0.23 / 0.00
June 14, 1996	346°	346°	1.97E-03	1.97E-03	0.21 / 0.21 / 0.00
August 7, 1997	347°	350°	1.95E-03	1.94E-03	0.21 / 0.21 / 0.01
August 6, 11, 1998	351°	350°	2.21E-03	2.03E-03	0.24 / 0.22 / 0.02
August 5, 9, 11, 1999	356°	3°	1.64E-03	2.01E-03	0.18 / 0.23 / 0.06
Average Values (Standard Deviation)	347° (± 5.0°)	349° (± 7.1°)	2.15E-03 (± 3.25E-04)	2.17E-03 (± 2.53E-04)	0.23 / 0.23 / 0.01

(b) RCRA monitoring well network: 299-W22-39, -W22-46, -W22-15.

Table 5. Trend-Surface Hydraulic Gradient and Groundwater-Flow Direction Determinations for Northern RCRA Wells Monitoring the WMA S Facility[(c)]

Date	Flow Direction, (0° = E; 90° = N)		Hydraulic Gradient (m/m)		Maximum Observed Difference, m: Water-Level Elevation/Total Head/Barometric Head
	Water-Level Elevation	Total Head	Water-Level Elevation	Total Head	
August 19-20, 1993	293°	293°	2.77E-03	2.77E-03	0.64 / 0.64 / 0.00
July 26, 1994	293°	293°	2.44E-03	2.44E-03	0.57 / 0.57 / 0.00
July 25, 27, 1995	301°	304°	1.88E-03	1.95E-03	0.47 / 0.50 / 0.03
June 13-14, 20, 1996	319°	325°	1.45E-03	1.58E-03	0.40 / 0.44 / 0.05
August 6-7, 1997	337°	343°	1.53E-03	1.50E-03	0.42 / 0.40 / 0.01
August 6, 12, 1998	347°	357°	1.76E-03	1.70E-03	0.46 / 0.41 / 0.05
August 9-10, 1999	354°	354°	1.85E-03	2.02E-03	0.44 / 0.50 / 0.04
Average Values (Standard Deviation)	321° (± 25.8°)	324° (± 27.9°)	1.95E-03 (± 4.81E-04)	1.99E-03 (± 4.67E-04)	0.49 / 0.49 / 0.03

(c) RCRA monitoring well network: 299-W22-44, -W22-45, -W22-13.