Development and Field Evaluation of an Aquifer Stratification Testing System

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Executive Summary

An aquifer stratification testing system was designed, constructed, and tested. The system was designed to characterize vertical profiles of both quantity and quality of water produced from actively pumping groundwater wells. During testing, all systems functioned properly and either met or exceeded anticipated performance requirements.

The system was first tested in an irrigation well completed in the Edwards aquifer and owned by the Onion Creek Club located in Manchaca, Texas, just south of Austin. The purpose of the exercise was to assess test systems performance and to develop and refine testing procedures. Dye-tracer injection tests were performed at five depths from 325 to 405 ft, and a velocity/discharge profile was generated for that interval. The profile was consistent with the karst nature of the Edwards aquifer. Water sampling procedures were tested and evaluated, but no samples were analyzed.

Further testing was performed in two public water supply wells completed in the Ogallala Formation of the High Plains aquifer and belonging to the City of Wolfforth, Texas, to investigate arsenic and fluoride concentration distributions, which both exceed maximum contaminant level (MCL) standards for arsenic (10 μg/L) and fluoride ((4 mg/L) in produced water. The results indicated vertical trends in both arsenic and fluoride aquifer concentration profiles, although concentrations at all depths exceeded MCL standards for both arsenic and fluoride. Despite distinctly different water discharge profiles between wells, a consistent inverse depth relationship between arsenic and fluoride concentrations was identified in both wells. Aquifer arsenic concentrations generally decreased, and aquifer fluoride concentrations generally increased with increasing depth. Aquifer arsenic concentrations ranged from 11.4 to 14.1 μg/L in Well 8 and from 16.0 to 22.2 μg/L in Well 5. Aquifer fluoride concentrations ranged from 5.53 to 7.98 mg/L in Well 8 and from 4.28 to 5.87 mg/L in Well 5.

Future system and procedural modifications should focus on improving and enhancing the flexibility and data acquisition capabilities of the system. Future field testing programs should be performed to develop a water quality stratification database to enhance general understanding of groundwater quality and flow processes and provide insight on potential contamination sources.

Introduction

Knowledge of the vertical distribution or stratification of aquifer water quality can provide extremely useful information for public water supply managers. The vertical distribution of water quality can provide insights into the mechanisms and processes controlling water quality and may provide information indicating the source of contaminants. In areas where contaminants are present in produced groundwater, knowledge of the flow and concentration profile within a well may provide producers options for contaminant mitigation by modifying either well construction or well management practices, as opposed to implementing expensive treatment options.

The aquifer stratification test system is a mobile test system designed to characterize water quality stratification in actively pumping groundwater wells. The test system consists of two major subsystems:

1. A dye-tracer injection and monitoring system.
2. A discrete depth well sampling system.

The complementary test subsystems are designed to identify both the quantity and quality of water produced over specific depth intervals. Both systems use small-diameter (≤~1 inch) down-hole components that require small access ports to the well casing. The small diameter both enhances the probability of system access to depths below the pump and limits the probability of becoming obstructed during emplacement or retrieval. A dye-tracer injection system is used to characterize the flow velocity profile. A small volume pulse of concentrated dye solution is injected at various depths within the flowing water column. Dye concentrations are monitored in the pump discharge flow stream using a high-precision fluorometer and automatically recorded using a data logger. The injection test results for different depths are combined to generate a flow velocity profile. The discrete depth sampling system is subsequently used to obtain water quality samples. Vertical aquifer water quality profiles for specific constituents are determined by combining the flow velocity test and water quality test results.

The mobile test system is based on a design originally developed by the U.S. Geological Survey, with several enhancements and modifications. The primary design enhancements of this system relative to the USGS system relate to the overall ease and precision of operation through the use of a centralized and enhanced process control system. A key component of the design enhancement is a control panel enclosure, which houses both electrical and pneumatic components that control and monitor most test procedure aspects. Once either test subsystem is positioned in the well, which generally requires a minimum of two personnel, a single operator can perform all testing, sampling, and data processing. A second enhancement incorporates an external data logger in the dye-tracer monitoring system, allowing greater precision of elapsed time measurement and extended flexibility of data acquisition, processing, and archival. A third enhancement adds a vacuum pump and associated plumbing system to the sampling system, which decreases both the time required to obtain water samples from shallow submergence depths and the amount of high-pressure gas consumed during the sampling process.

The systems are mounted in an 8- × 14-ft enclosed trailer. Two electrically powered hose reels provide storage for the hose assemblies used in the testing methods. Power is provide by an onboard generator or by external 120-volt AC if available at the wellhead. A 12-volt charging and supply system with a 100-amp-hour storage capacity provides DC power, which applies to most of the test system control and monitoring system requirements. All power circuits are overload-protected by circuit breakers (AC) or fuses (DC), and the AC circuits additionally employ ground fault circuit interrupter (GFCI) devices to enhance operator safety.

Dye-Tracer Injection and Monitoring Methods

The water velocity profile of a groundwater well depends on several factors, including depth position and discharge rate of the pump, pipe diameter, well construction, and possible variability of aquifer hydraulic properties with depth. Dye-tracer testing measures the average flow velocity between tested depths, from which estimates of the cumulative total well flow and interval average aquifer flow can be calculated.

The dye-tracer injection system consists of six major components: (1) a 5-gallon-capacity dye-tracer holding tank, (2) a high-pressure piston pump to circulate and inject the dye, (3) a relief (unloader) valve to relieve excessive injection pressures, (4) an electro-pneumatically actuated crossover valve to control fluid flow states, (5) a timer relay to control the duration of injection, and (6) an injection hose assembly to deliver the dye tracer to the well (Figure 1).

A dye-tracer pulse injection cycle is initiated by first turning on the pump. Upon power-up, the flow path is in a default low-pressure circulation state between the pump and the reservoir tank. Tracer injection is initiated with a push button located on the control panel, which causes a relay to switch the crossover valve into a high-pressure injection state, and the relay timer begins a countdown for a preprogrammed duration (Figure 1a). A pressure wave propagates down the hose assembly, and the dye-tracer pulse is released into the well through a terminal check valve. When the timer countdown reaches zero, the relay is turned off, the crossover valve reverts to the low-pressure circulation state (Figure 1b), and the pressure in the injection hose assembly is relieved by backflow to the reservoir tank.

The dye tracer used is (nontoxic) Rhodamine WT, which is commonly used in studies to characterize both surface and groundwater flow. The U.S. Environmental Protection Agency has recommended a concentration of 10 μg/L (ppb) for water entering a treatment plant (prior to treatment and distribution) and of 0.1 μg/L (ppb) for drinking water. A dye solution concentration of approximately 200 mg/L (ppm) and an approximate pulse injection volume from 5 to 10 cm^3 was used during field evaluation testing. Peak wellhead concentrations ranged between 0.5 and 70 μg/L (ppb), indicating approximately four to five orders of magnitude dilution over travel distances within the tested wells, which ranged from 220 to 675 ft. At these dilution rates, dye concentration in the produced water would be undetectable (<0.01 μg/L) after no more than 500 ft of travel distance in downstream conveyance plumbing.

The dye-tracer monitoring system consists of a fluorometer and a data logger. The well pump stream is monitored in real time for dye-tracer concentrations using a temperature-compensated fluorometer (Model 10-AU, Turner Designs, Sunnyvale, CA). Water flow is diverted from the wellhead using a standard hose bib connection and garden hose routed to the fluorometer.

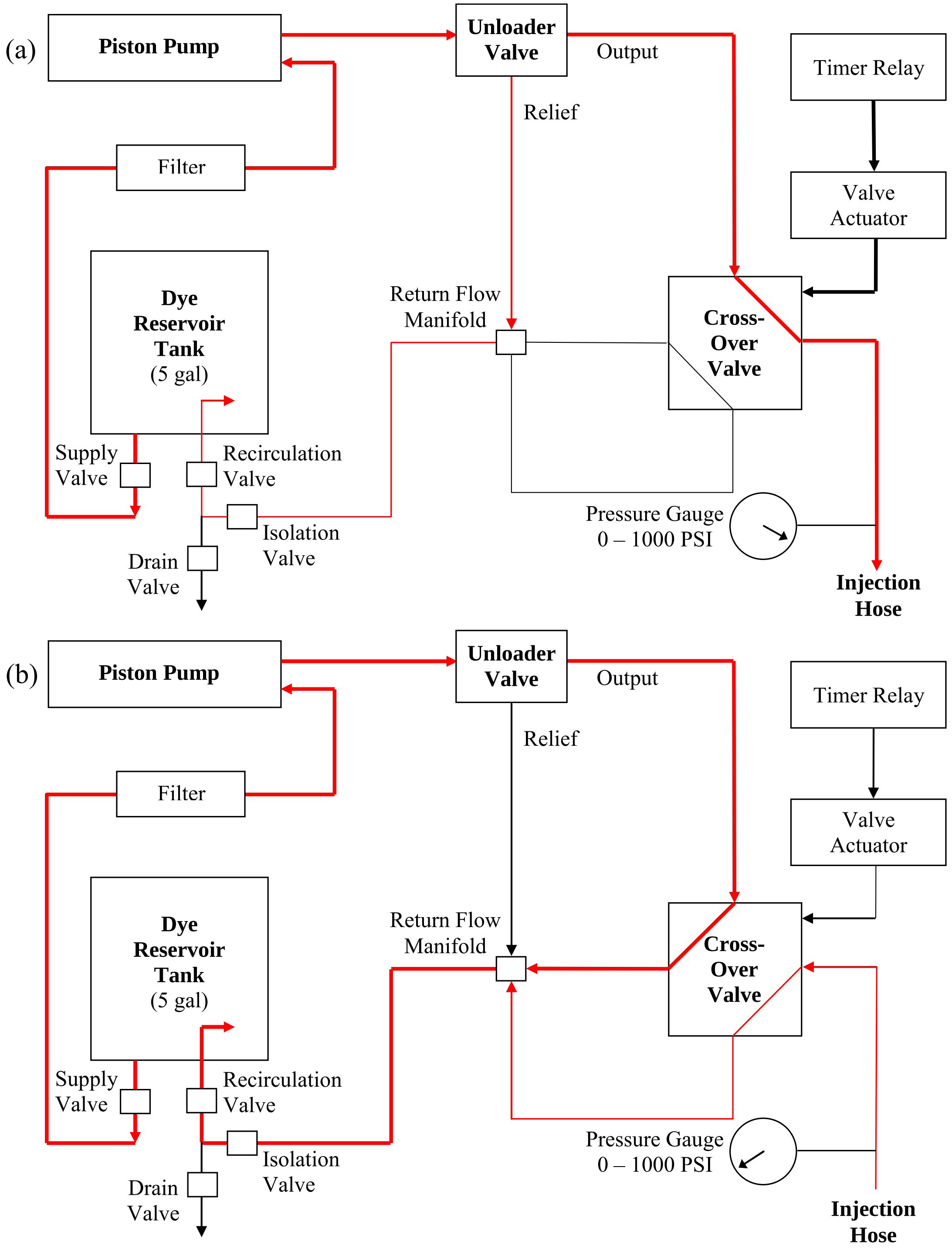

Figure 1: Dye-tracer injection system schematic showing (a) high-pressure injection and (b) low-pressure circulation states. Heavy red lines indicate primary dye-tracer fluid flow paths. Narrow red lines in a show flow return path from unloader valve for excess injection pressure. Narrow red lines in b show pressure release return flow path from hose following injection.

The fluorometer is equipped with a continuous flow cell and is calibrated for the range from 0 to 100 μg/L (ppb) with a sensitivity of 0.01 μg/L. This range represents the linear range of fluorescent response for Rhodamine WT concentrations using this instrument configuration. The fluorometer output signal is connected to an external data logger (CR10X, Campbell Scientific, Logan, UT). (The fluorometer also has an internal data logger that is used as a backup.) Logging by the external data logger is activated by the same push button that initiates injection cycles, ensuring precise measurement of elapsed test time. Both the elapsed test time and fluorometer output signal are digitally recorded at 1-s intervals and graphically displayed in real time.

Discrete Depth Water Quality Sampling Methods

The water quality profile of a groundwater well depends on both the discharge profile and the vertical variability of water quality in the aquifer. Discrete depth samples provide cumulative constituent flux concentrations as the water in the well flows toward the pump inlet. These data are combined with the dye-tracer velocity/discharge profile to estimate average constituent concentrations entering the well between tested depths.

The discrete depth sampling system consists of five major components: (1) a dual-tube hose that delivers regulated pressures into the well and returns sample water from the well; (2) a passive pump connected to the sampling hose; (3) a high-pressure gas source that provides energy to drive the sampling process; (4) a system of regulators, valves, pressure transducers, and pressure displays to control and monitor the sampling process; and (5) sample processing equipment to collect, filter, and store the samples.

The high-pressure source consists of either an argon or nitrogen gas tank with an initial storage capacity of approximately 150 ft^3 at 2,200 PSI. From two to three such tanks can be safely stored and transported in the sampler trailer. The sampling process progresses as an alternating series of pressurization and venting cycles that move sample water through the passive pump and sample hose tubes (Figure 2). Water initially enters both tubes driven by hydrostatic pressure at the sample pump. A pressure cycle then drives water down the pressure tube and up the sample tube, while the lower pump check valve prevents backflow of sample into the well. The subsequent vent cycle releases the compressed gas introduced during the pressure cycle, and the hydrostatic pressure at the pump again forces new sample water into the pressure tube. During the vent cycle, backflow of water stored in the sample tube is prevented by the upper pump check valve because the hydrostatic pressure in the sample tube is now greater than the hydrostatic pressure at the sample depth in the well. As constructed, the sampling system is capable of pressures up to 450 PSI, equivalent to a sampling depth of approximately 1,000 ft. The length of the (replaceable) sample hose currently in use is 600 ft.

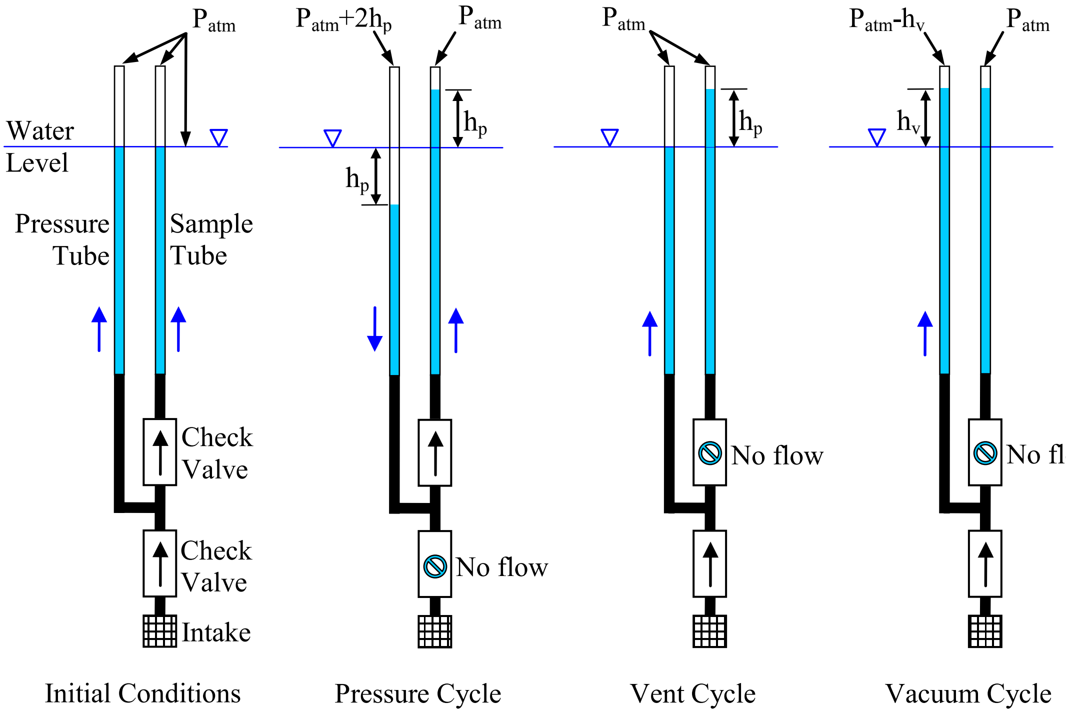

Figure 2: Discrete depth sampling system schematic. Sample hose and sample pump conditions shown in response to various sample cycle conditions. Arrows indicate the direction of sample flow.

The process of "stacking" water in the sample tube through alternate pressure-vent cycle pairs is controlled by the control panel valves. The pressures required for sequential sample stacking cycles are calculated from the depth to water, the sample submergence depth (defined as the depth of the sample pump intake below the well water level), and the height of water stored in the sample tube from previous cycles. Cycles are repeated until there is sufficient sample volume for the intended purpose, at which point the entire stacked water column is forced up and out of the sample tube by overpressuring the pressure hose. The storage capacity of the pressure and sample tubes is approximately 2.4 mL/ft.

Under shallow sample pump submergence depths (< ~50–60 ft), where both hydrostatic pressure gradients and cycle sample volumes are relatively small, vacuum-vent cycle pairs can be employed during the sampling process to increase sample volume recovery per cycle (Figure 2). Additional sample water is drawn into the system using a vacuum pump, which effectively raises the water level in the well. Employed prior to the first pressure cycle, a vacuum cycle draws sample water into both the pressure and sample tubes. Subsequent sample cycles draw water into only the pressure hose. With the vacuum pump currently in use, water can be pulled approximately 20 ft above the water level. The use of vacuum cycles also has the advantage of reducing the amount of compressed gas consumed per volume of sample produced. Vacuum cycles may also be employed during deeper submergence conditions to speed pressure-tube water level recovery time following vent cycles, which slows during the final stages as the pressure-tube water level approaches the well water level and the pressure gradient driving the flow declines.

Data Processing Methods

Data processing of stratification test data requires integrating well velocity/discharge results from the dye-tracer injection tests with the constituent concentration analysis results from discrete depth water samples.

The total mass of dye, D^T, recovered during a tracer test is determined by integrating the total well discharge, Q^T, and tracer concentration, C, over time, t. Assuming that both Q^T and the concentration measurement time interval, Δt, are constant during the test period:

$$D^T = \int Q^T C dt = Q^T \int C dt = Q^T \Delta t \sum C \qquad \text{eq. 1}$$

The value of D^T is useful in examining consistency between tracer test injection volumes. The dye-tracer center-of-mass arrival time is used to determine the average flow velocity between tested depths.

The first-arrival time of dye is identified as the first data record at which a consistent increase above background concentration occurs. A running cumulative sum of the concentration measurements is calculated beginning at the first-arrival record and across all subsequent records to the point where concentration returns to background level.

Under the same assumptions of constant Q^T and Δt, the center-of-mass arrival time, t^m, is determined as the elapsed test time at which the cumulative sum of dye concentrations represents 50% of the total cumulative sum:

$$t^m = \frac{\sum C}{\sum C_{total}} = 0.50 \qquad \text{eq. 2}$$

The average flow velocity, υ^a, over a given depth interval, i, is calculated as the absolute difference between the bounding test interval depths z_1 (closest to the pump) and z_2 (farthest from the pump) divided by the difference between the respective center-of-mass arrival times:

$$\upsilon_i^a = \frac{|z_2 - z_1|}{t_2^m - t_1^m} \qquad \text{eq. 3}$$

The cumulative well discharge, Q^c, is estimated as an average over interval i, from the interval average flow velocity, and the interval average cross-sectional area:

$$Q_i^c = \upsilon_i^a \pi r_i^2 \qquad \text{eq. 4}$$

The interval average cross-sectional area, πr^2, within the well casing radius, r_c, must be adjusted for displacement resulting from the sum of obstructions, r_o, due to riser pipes, electrical cables, etc., that are present between the injection depths:

$$r_i^2 = r_c^2 - \sum r_o^2 \qquad \text{eq. 5}$$

The interval average aquifer discharge, Q^a, is estimated as the difference between the cumulative well discharges for the tested interval i and the interval i-1 next farthest from the pump:

$$Q_i^a = Q_i^c - Q_{i-1}^c \qquad \text{eq. 6}$$

Discrete depth samples provide a constituent flux concentration, C^f, in the well water at a given depth, z. The constituent average aquifer-flux concentration, C^a, flowing into the well over the depth interval i between depths z_1 (closest to the pump) and z_2 (farthest from the pump) is estimated as the difference between the products of the cumulative well discharge estimates and the interval constituent well flux concentrations divided by the interval average aquifer discharge:

$$C_i^a = \frac{Q_i^c C_{z_1}^f - Q_{i-1}^c C_{z_2}^f}{Q_i^a} \qquad \text{eq. 7}$$

The units of discharge cancel out in eq. 7. Thus, aquifer-flux concentration calculations may be performed by substituting average discharge with either average velocity or percentage of total average velocity measurements, provided that the cross-sectional flow area remains constant throughout the tested depth intervals.

Field Evaluation Tests

We performed field testing of the system on three wells at two field locations. The first tests were performed June 27–28, 2006, on a single irrigation well located in Manchaca, Texas, owned by the Onion Creek Club just south of Austin. The purpose of that testing was to investigate and benchmark the performance of the test systems and to develop and refine testing procedures. The Onion Creek Club well is an open-hole completion installed in the Edwards aquifer between 222 and 490 ft depth. Pumping rate during testing was 211 gpm. Further testing was performed July 29-August 3, 2006, on two public water supply wells owned by the City of Wolfforth, Texas, just southwest of Lubbock. The City of Wolfforth wells are cased wells installed in the Ogallala Formation of the High Plains aquifer to depths ranging from 195 to 213 ft depth. Pumping rates during testing ranged from 25.5 to 64 gpm. Similar to groundwater from other public water supply systems in the region, groundwater produced from the City of Wolfforth wells have concentrations of arsenic and fluoride that exceed EPA drinking water standards.

Velocity profiles were characterized for all wells using the dye-tracer injection system. Water samples were collected from all wells using the discrete depth sampling system, but only samples from the public water supply wells were analyzed.

Onion Creek Club Irrigation Well

The Onion Creek Club irrigation well tested is one of three wells that supply irrigation water for the club's golf course (Table 1). Water is pumped from the wells to holding ponds. The wells are surface cased to the top of the Edwards aquifer, which is located at approximately 220 ft depth. Total well depths are approximately 500 ft. The well tested (58-50-836) is open hole completed with a 7.875 in diameter to 490 ft. The depth of the pump was unknown during testing.

Table 1: Onion Creek Club irrigation well physical characteristics.

Parameter	*Value*	*Units*
Well depth	490	ft
Pump intake depth	> 405	ft
Open-hole interval	222 to 490	ft
Diameter	7.875	inches
Test pump rate	211	gpm
Test depth to water	227	ft

Stratification Testing Results

Testing was performed June 27–28, 2006. Total well discharge was monitored during testing using an existing totalizing flow meter installed near the wellhead. Dye-tracer injection tests were performed at five depths (Figure 3). From two to three tests were conducted at each depth, except for the 325-ft depth, where only one test was performed. Center-of-mass arrival time variability ranged from 0.3 to 1.0% and averaged 0.6% (Table 2). Peak dye concentrations at the wellhead ranged from 2.7 to 21.2 ppb (Figure 3). Dye-tracer recovery mass values were consistent between all tests at and below 350 ft depth. The lower mass recovery at 325 ft was due to a smaller injection pulse volume resulting from a shorter injection time cycle. Mass recovery results indicate that the system performed repeat tests well.

Table 2: Center-of-mass arrival times (seconds) for repeated dye-tracer tests conducted at different depths in Onion Creek Club irrigation well. CV: coefficient of variance.

Depth (ft)	*Test 1*	*Test 2*	*Test 3*	*Average*	*CV*
325	986	–	–	986	–
350	717	721	–	719	0.4%
375	434	443	437	438	1.0%
400	249	247	246	247	0.6%
405	231	230	–	231	0.3%

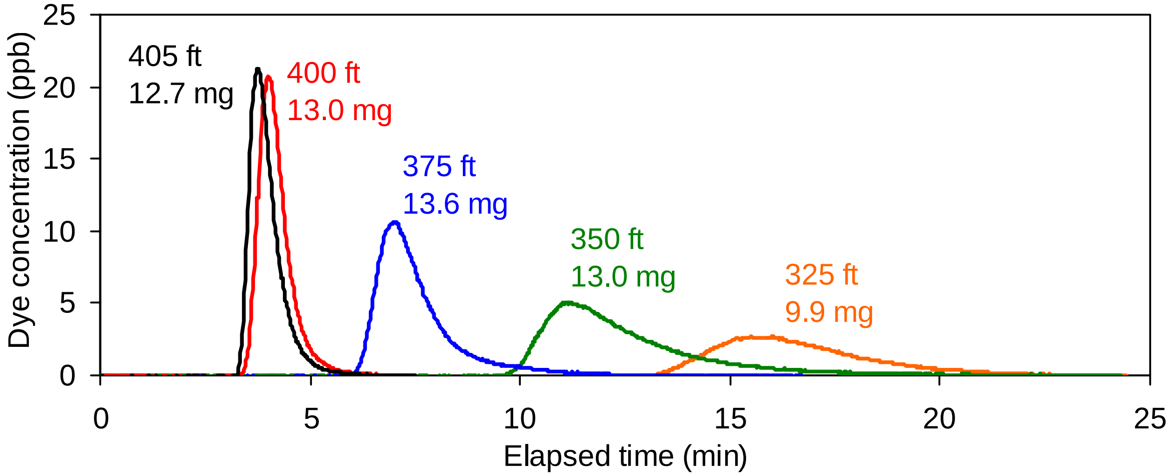

Figure 3: Dye-tracer breakthrough curves measured in well discharge for tests conducted at various depths in the Onion Creek Club well. The total dye mass recovered during each test is also shown.

Calculated interval average well velocities decreased slightly in the direction toward the pump between the two shallowest tested depths, which indicates that water may have actually been lost from the well between the 350 and 375 ft depths (Figure 4a). The velocity difference is small, however, and the indicated loss may be the result of measurement error or the result of slight variations in the (open) borehole diameter. Total flow originating from depths shallower than 400 ft was estimated to be 16.0 gpm, which represents approximately 7.6% of the total well discharge (211 gpm) during the test (Figure 4b). A large increase in flow was measured between the 400- and 405-ft depths, where flow velocity increased dramatically and approximately 9.6% of total well discharge originated. The measured aquifer discharge profile (Figure 4c), consistent with the karstic nature of the Edwards aquifer, indicates that approximately 83% of total well production originated from depths below 405 ft.

Water samples were collected to benchmark discrete depth sampling performance, primarily to determine that the procedures resulted in produced sample volumes in the anticipated range. Tests confirmed that each sampler production cycle produces approximately 2.4 mL of sample per foot of sample pump submergence in the well water column. Cycle procedures successfully produced sample without entrained compressed gas. The use of a vacuum pump to draw additional sample into the system under shallow submergence conditions enhanced recovered sample volumes.

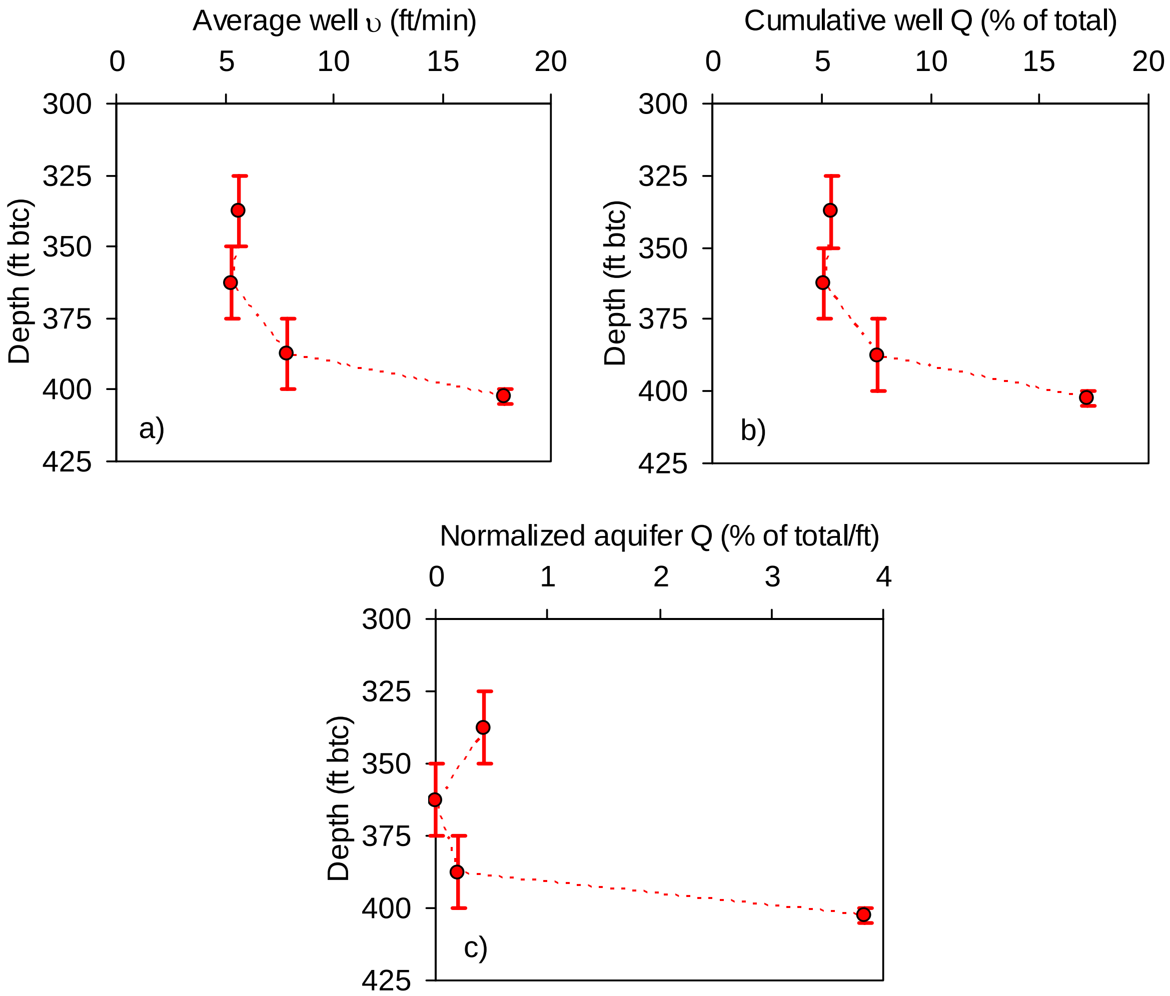

Figure 4: Results for (a) average well water velocities, (b) cumulative percent of total well discharge, and (c) normalized aquifer discharge calculations based on the average dye-tracer center-of-mass arrival times shown in Table 1. Points represent average values, whereas vertical error bars represent tested interval depth ranges.

City of Wolfforth PWS Wells

The City of Wolfforth supplies drinking water to approximately 2,500 residents with a network of 10 groundwater wells completed in the Ogallala Formation of the High Plains aquifer. Well depths range from 170 to 213 ft and average 200 ft. Many of the wells were originally installed for agricultural irrigation purposes and have been converted to public supply use. Current depths to (static) water level range from approximately 135 to 155 ft, and saturated thickness is estimated to range from 50 to 60 ft. Normal pumping rates generally range from 30 to 150 gpm. Groundwater quality data for well samples collected between 2001 and 2005 show arsenic concentrations ranging from 10 to 19.3 mg/L, with a median of 13.3 mg/L (25 samples), and fluoride concentrations ranging from 4.00 to 9.06 mg/L, with a median of 5.30 mg/L (93 samples) (TCEQ Public Water Supply Database). Total dissolved solids (TDS) over the same period ranged from 461 to 1,202 mg/L, with a median of 710 mg/L (28 samples).

Table 3: City of Wolfforth PWS well physical characteristics.

Parameter	*Well 8*	*Well 5*	*Units*
Well depth	195	213	ft
Pump intake depth	190	205	ft
Screened interval	115 to 195	148 to 203	ft
Static water level	142 (2004)	N/A	ft
Casing diameter	16	10.75	inches
Test pump rate	64	25.5	gpm
Test depth to water	160	163	ft

We performed tests on two of the public water supply wells to investigate potential water quality stratification with regard to both arsenic and fluoride concentrations. Testing was performed on Well 8 from July 29 through August 1, 2006 and on Well 5 from August 1 through 3, 2006 (Table 3). In the wells tested, the water column height ranged from 30 to 40 ft above the pump depth during testing. Total well discharge was monitored throughout testing using existing totalizing flow meters installed at the wellhead. The City of Wolfforth treats their produced water using metered chlorine gas injected into each well. Injection systems were turned off, and three well volumes were discharged prior to sampling each well. During water quality sampling, parameters including temperature, pH, and specific conductance of the total well discharge were continuously monitored and found to be sufficiently stable pursuant to Texas Water Development Board (TWDB) groundwater sampling guideline requirements.

Stratification Testing Results

The dye-tracer tests were performed at four depths in Well 8 and at six depths in Well 5 (Figure 5). Two to three tests were conducted at each depth. Center-of-mass arrival time variability ranged from 0.3 to 1.8% and averaged 1.0% (Tables 4 and 5).

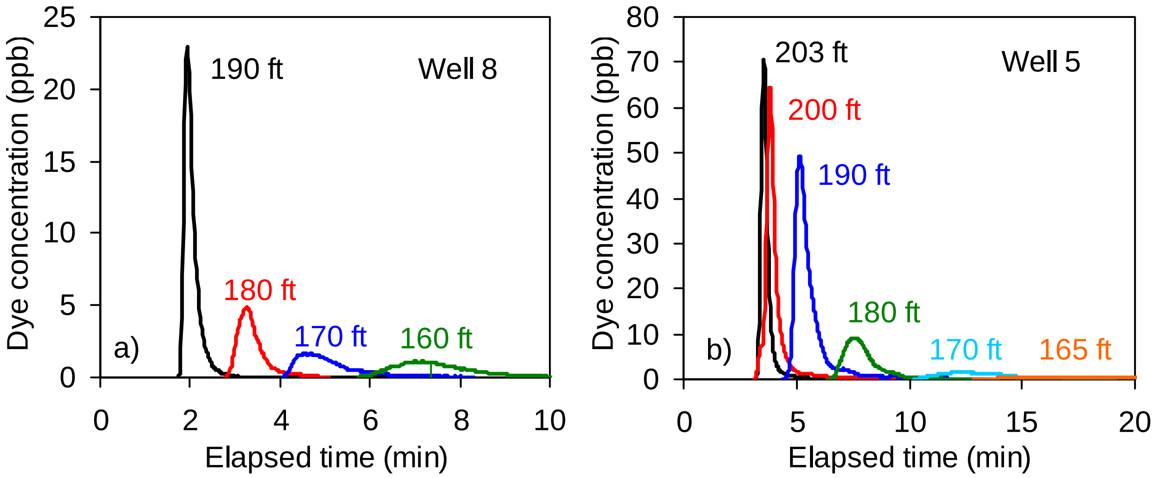

Figure 5: Dye-tracer breakthrough curves measured in the well discharge for tests conducted at various depths in (a) Well 8 and (b) Well 5.

Table 4: Center-of-mass arrival times (seconds) for repeated dye-tracer tests conducted at different depths in City of Wolfforth PWS Well 8. CV: coefficient of variance.

Depth (ft)	*Test 1*	*Test 2*	*Test 3*	*Average*	*CV*
160	443	440	446	443	0.7%
170	293	295	292	293	0.5%
180	194	196	198	196	1.0%
190	117	114	114	115	1.8%

Table 5: Center-of-mass arrival times (seconds) for repeated dye-tracer tests conducted at different depths in City of Wolfforth PWS Well 5. CV: coefficient of variance.

Depth (ft)	*Test 1*	*Test 2*	*Test 3*	*Average*	*CV*
165	1015	1037	-	1026	1.5%
170	776	778	-	777	0.2%
180	466	466	475	469	1.1%
190	314	326	320	320	1.8%
200	236	233	234	234	0.6%
203	215	214	215	215	0.3%

Calculated interval average well velocities increased between tested intervals in the direction of the pump, which indicates that water entered the well over all tested depths

(Figure 6a). Aquifer discharge profiles showed distinct differences between the two wells (Figure 6b). Aquifer discharge into Well 8 displayed an upward increasing trend, whereas discharge into Well 5 showed a downward increasing trend (Figure 6c). In Well 8, an estimated 83% of total discharge originated from depths above 180 ft., whereas in Well 5, an estimated 74% of total discharge originated from depths below 180 ft.

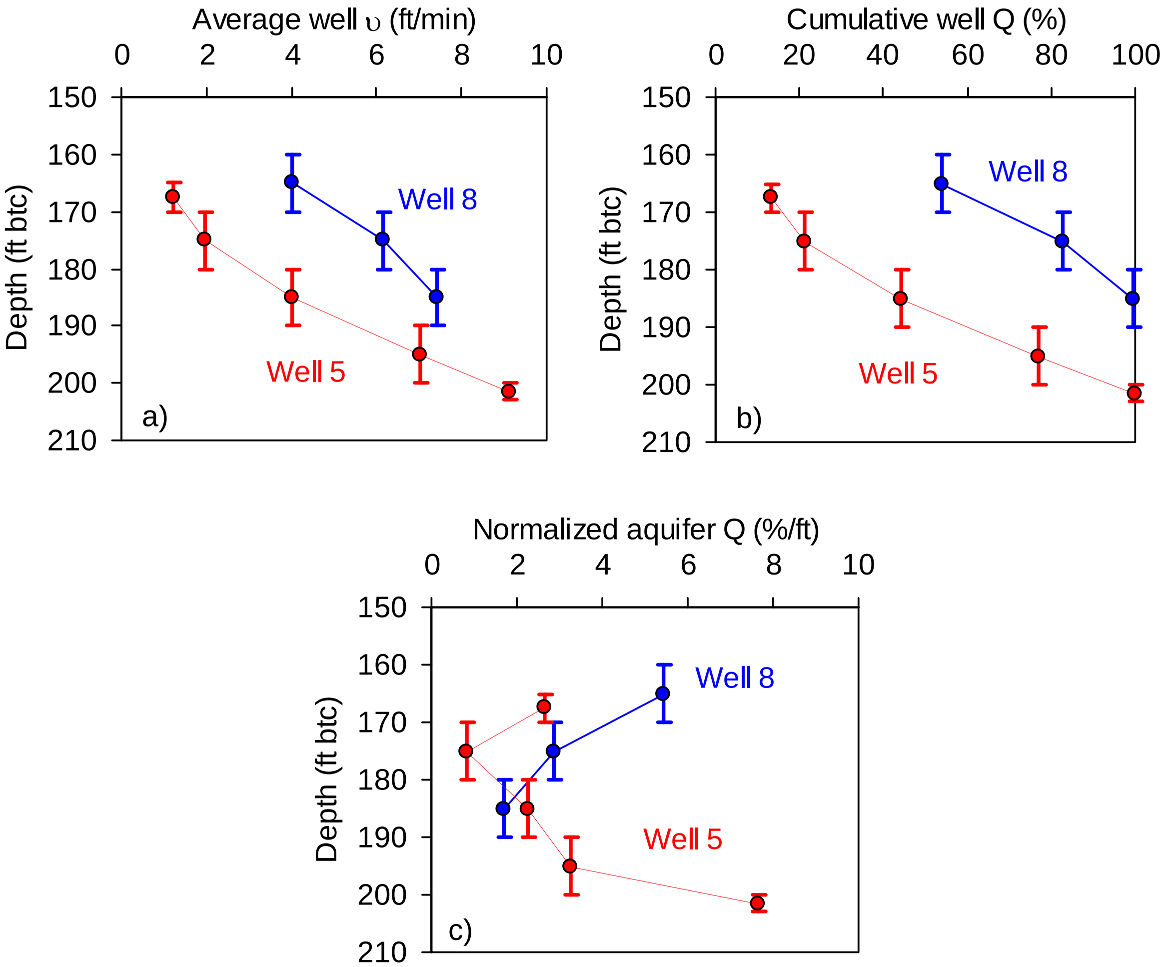

Figure 6: Results for (a) average well water velocities, (b) cumulative percent of well discharge, and (c) normalized aquifer discharge calculations based on the average dye-tracer center-of-mass arrival times shown in Tables 1 and 2. Points represent average values, whereas vertical error bars represent tested interval depth ranges.

Water parameters in the pump discharge flow stream were continuously monitored during discrete depth sampling, and parameter values were similar for the two wells (Table 6), except for dissolved oxygen (DO) and oxidation-reduction potential (ORP), which showed significant differences. These differences are consistent with the aquifer discharge profiles of the two wells: Well 8 received more of its total discharge from shallower depths and has higher DO and ORP than Well 5.

Discrete depth water samples were analyzed for arsenic, vanadium, fluoride, chloride, and sulfate concentrations. Results showed distinct differences in constituent concentrations between the two wells (Table 7). Well 8 had generally lower concentrations of arsenic, vanadium, chloride, and sulfate and higher concentrations of fluoride than Well 5. All discrete depth sample concentrations for arsenic and fluoride exceed MCL concentrations.

Table 6: Total discharge water parameters monitored during discrete depth sampling.

Parameter	*Well 8*	*Well 5*	*Units*
Temperature	19.4	19.8	C
pH	7.31	7.20	-
Specific conductance	1147	1271	µS/cm
Salinity	0.57	0.62	PSS
Dissolved oxygen	5.23	0.35	mg/L
Dissolved oxygen	67.7	4.8	% saturation
Oxidation-reduction potential	840	558	mV

Table 7: Discrete depth sample analysis results.

	Depth (ft)	*Arsenic (µg/L)*	*Vanadium (µg/L)*	*Fluoride (mg/L)*	*Chloride (mg/L)*	*Sulfate (mg/L)*
Well 8	160	14.0	57.0	5.39	141	130
	170	14.1	57.7	5.53	136	127
	180	13.8	56.1	5.28	128	122
	190	13.4	56.6	5.74	124	126
Well 5	170	22.2	153	4.28	206	144
	190	19.3	130	4.60	171	142
	200	17.9	106	5.14	134	127
	203	18.8	107	4.93	140	131

Table 8: Estimated aquifer-flux concentrations over tested depth intervals.

	Top Depth (ft)	*Bottom Depth (ft)*	*Arsenic (µg/L)*	*Vanadium (µg/L)*	*Fluoride (mg/L)*	*Chloride (mg/L)*	*Sulfate (mg/L)*
Well 8	160	170	14.1	57.7	5.53	136	127
	170	180	13.2	53.1	4.81	113	113
	180	190	11.4	59.0	7.98	104	146
Well 5	165	170	22.2	153	4.28	206	144
	170	190	18.1	120.2	4.74	156	141
	190	200	16.0	73.7	5.87	84	107
	200	203	21.8	110.3	4.23	160	144

Results of the aquifer-flux concentration profile calculations indicate vertical trends in water chemistry (Table 8, Figures 7 and 8). Despite the distinct difference between aquifer discharge profiles, both wells showed similar trends for most analyzed constituents. With increasing depth, arsenic, vanadium, and chloride concentrations decreased, fluoride concentrations increased, and sulfate concentrations were variable.

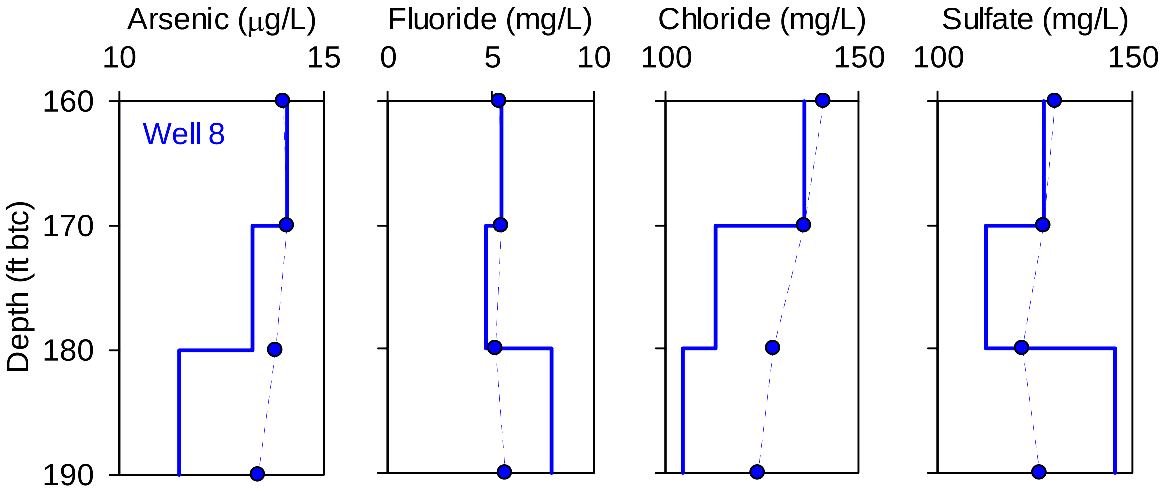

Figure 7: Constituent concentration profiles for Well 8. Points connected by dashed lines represent well water flux concentrations in discrete depth samples, whereas heavy lines represent calculated average aquifer-flux concentrations.

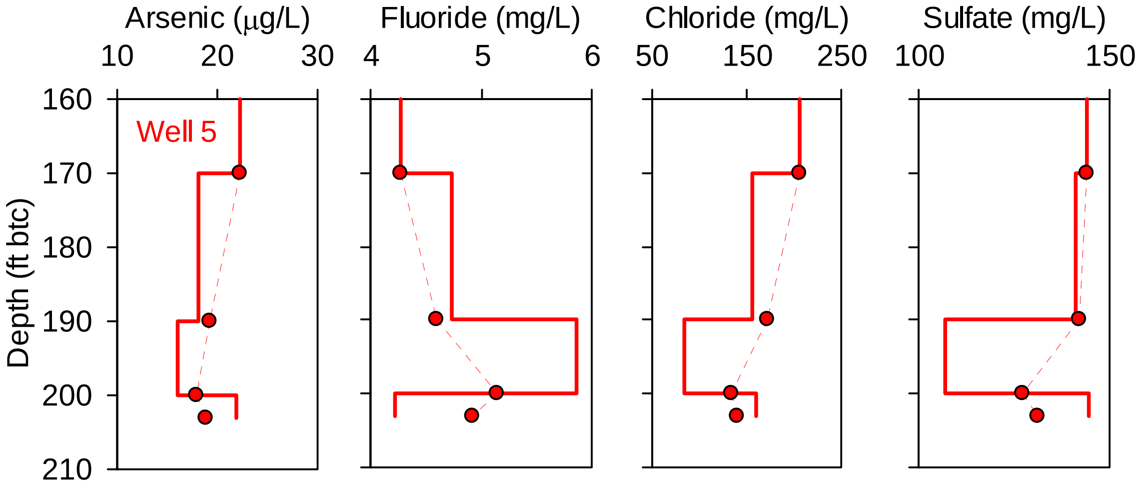

Figure 8: Constituent concentration profiles for Well 5. Points connected by dashed lines represent well water flux concentrations in discrete depth samples, whereas heavy lines represent calculated average aquifer-flux concentrations.

A distinct reversal of these trends is noted at the base of the Well 5 profile over the 200- to 203-ft-depth interval (Figure 8), where aquifer discharge to the well was measured at the highest rate and an estimated 23% of total well discharge originated. The interval has a vertical separation of only 3 ft and is located in close proximity to the pump intake where higher flow velocities exist. These factors result in increased sensitivity of the velocity calculations to small variations in the dye center-of-mass arrival times. As a check on validity of the vertical concentration profile observations already stated, the bottom two tested intervals were combined into a single interval from 190 to 203 ft depth, and a parallel analysis was performed (Figure 9). The resulting average aquifer-flux concentration of the combined intervals provides vertical concentration profiles that remain consistent with the original analysis observations—that arsenic, vanadium, chloride, and sulfate concentrations decrease with increasing depth and fluoride concentrations increase with depth.

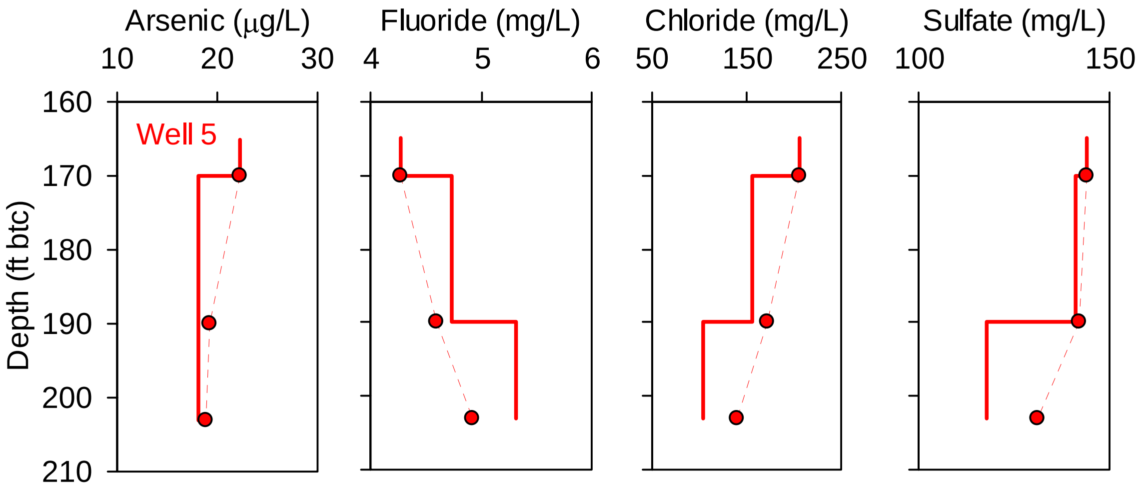

Figure 9: Constituent concentration profiles for Well 5 that combine lowest two tested intervals. Points connected by dashed lines represent well water flux concentrations in discrete depth samples, whereas heavy lines represent calculated average aquifer-flux concentrations.

The aquifer-flux concentration estimates for Wells 8 and 5 were compared with limited historical well sample data for City of Wolfforth wells from both the TWDB and the TCEQ PWS databases, which represent composite (i.e., total depth- and flow-integrated) groundwater samples (Figure 10). Power law regression models were fitted separately to the composite and aquifer-flux sample data and compared. Correlation between arsenic and fluoride in the composite samples is not indicated, whereas aquifer-flux concentrations show strong negative correlation. Aquifer-flux correlations between both fluoride/chloride and arsenic/chloride are similar to, but generally more pronounced than, historical composite sample correlations. Chloride concentrations are strongly correlated with TDS concentrations and provide an indication of general water quality. The generally lower correlations between historical data may be attributed partly to temporal variability of water quality, differences between sampling or analytical methodologies, or a combination thereof.

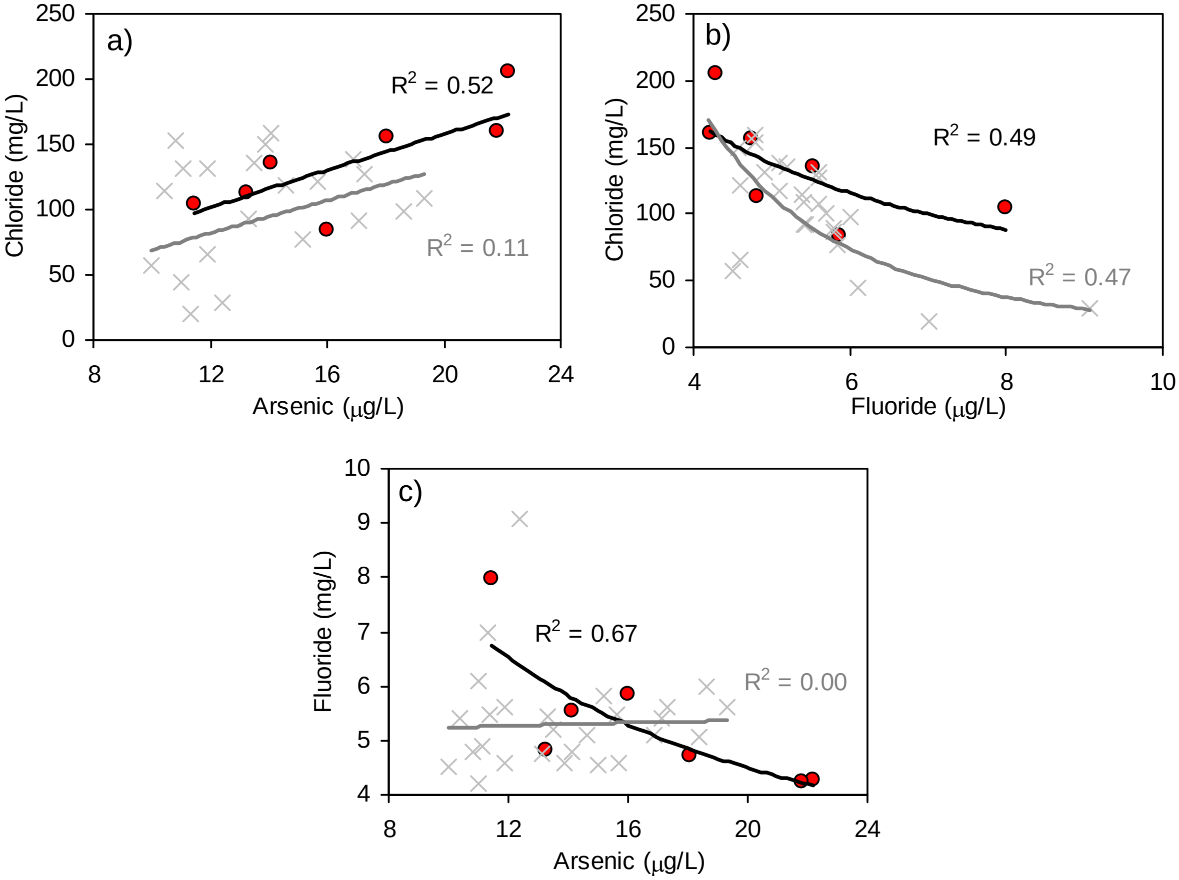

Figure 10: Relationships between historical concentrations of arsenic, fluoride, and chloride in City of Wolfforth PWS wells (gray points) and aquifer-flux concentrations from Wells 8 and 5 (red points). Gray lines and R^2 values represent historical sample regressions, and black lines and R^2 values represent aquifer-flux regressions with power law models.

Local aquifer-flux concentration estimates from Wells 8 and 5 were compared with regional groundwater well sample data in the TWDB database for wells completed in the Ogallala aquifer in Texas (Figures 11 to 13). Whereas all of the local data fall within the range of regional data distributions, some of the local data display distinctly different trends relative to regional trends. In the regional distribution, arsenic and fluoride appear to have a generally positive correlation, whereas the local data suggest a much stronger negative correlation (Figure 11). The regional and highly positive correlation between arsenic and vanadium is reflected in the local data (Figure 11).

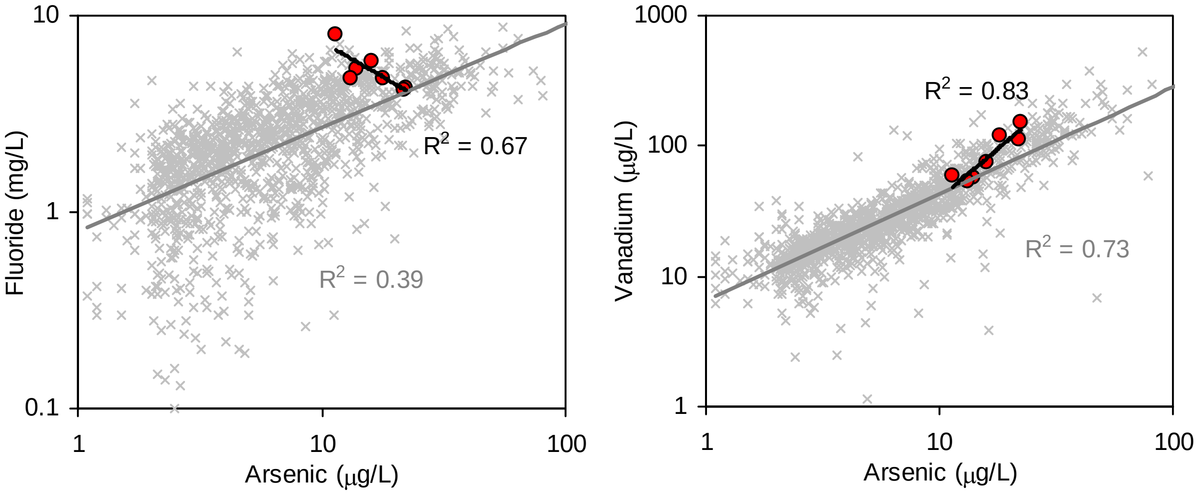

Figure 11: Relationships between arsenic, fluoride, and vanadium in regional wells (gray points) and local aquifer-flux concentrations from Wells 8 and 5 (red points). Gray text R^2 values represent regional regression results, and black text R^2 values represent local regression results for power law models.

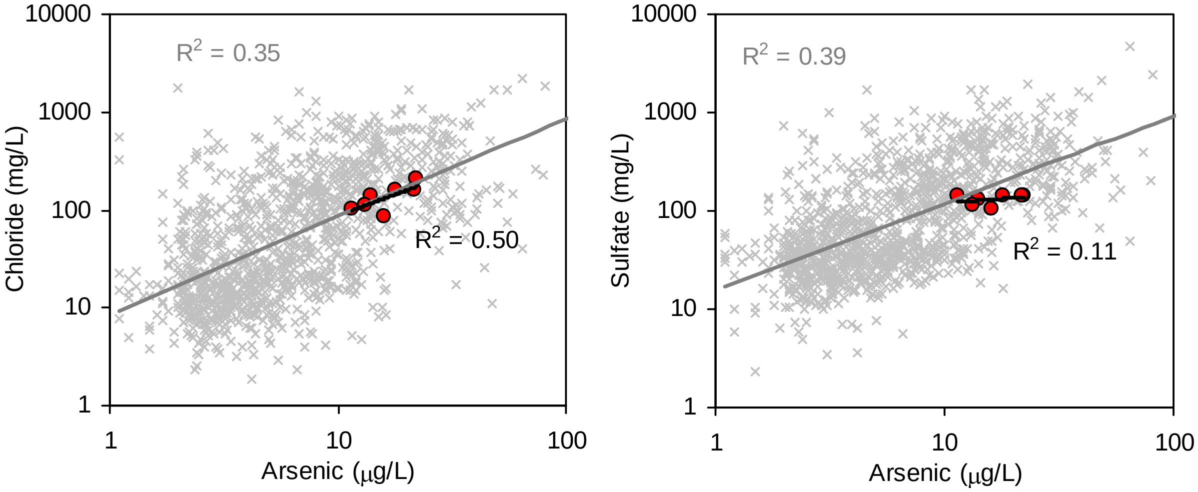

Figure 12: Relationships between arsenic, chloride, and sulfate in regional wells (gray points) and local aquifer-flux concentrations from Wells 8 and 5 (red points). Gray text R^2 values represent regional regression results, and black text R^2 values represent local regression results for power law models.

Generally positive correlations between both arsenic/chloride and arsenic/sulfate are indicated on the regional scale, whereas arsenic shows a substantially greater positive correlation with chloride but no correlation with sulfate on the local scale (Figure 12). Finally, fluoride shows very weak positive correlations with both chloride and sulfate on the regional scale, whereas fluoride/chloride are negatively correlated and fluoride/sulfate show no correlation on the local scale (Figure 13).

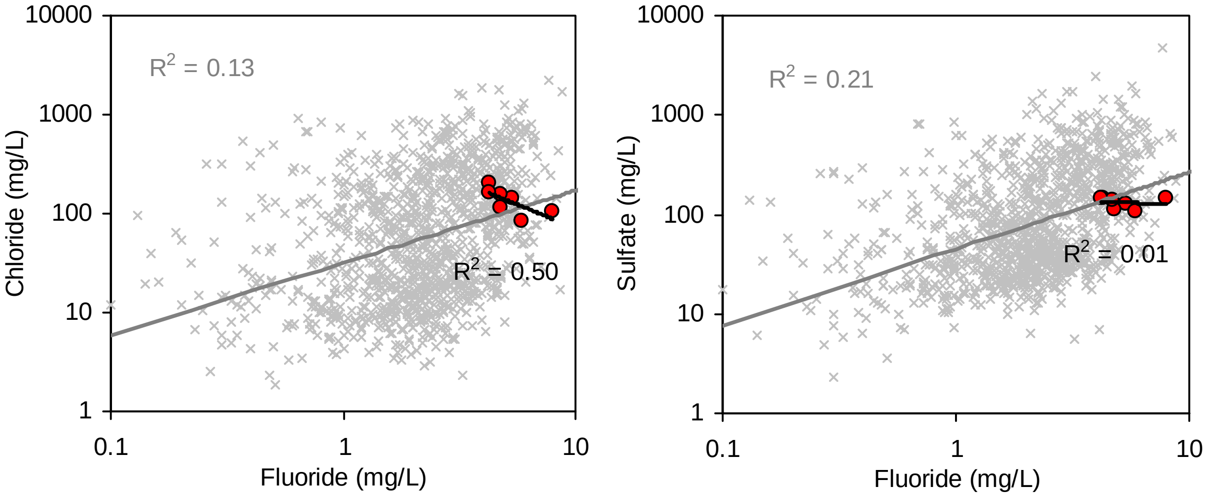

Figure 13: Relationships between fluoride, chloride, and sulfate in regional wells (gray points) and local aquifer-flux concentrations from Wells 8 and 5 (red points). Gray text R^2 values represent regional regression results, and black text R^2 values represent local regression results for power law models.

Comparisons of correlations between limited local aquifer-flux concentrations with both local and regional composite sample concentrations indicate that composite sampling may provide results that obscure the relationships and processes that control the distribution of some dissolved species on a local scale. Future work should focus on expanding the application of stratification sampling to determine the spatial distribution and temporal persistence of stratification found in this study.

City of Wolfforth PWS Recommendations

Given that none of the tested depth intervals produced water with either arsenic or fluoride concentrations lower than MCL values, neither of the tested wells seems a potential candidate for modification. The generally high concentrations of both arsenic and fluoride in all network wells identified through historical composite groundwater samples, coupled with the inverse depth relationship between arsenic and fluoride concentrations identified in this study, suggests that a lower arsenic concentration zone is likely to have higher fluoride concentration, and vise-versa, resulting in offsetting benefits. However, mitigation efforts that concentrate on just one of the contaminants may be possible. For example, overall arsenic concentration might be reduced by selective water production from deeper depth intervals.

System Performance Review

The field-evaluation phase of system development was very successful. The system performance met or exceeded anticipated requirements. The times required to perform different testing procedures decreased as testing progressed and operator experience increased. The two testing procedures, velocity profiling and discrete depth sampling, each required approximately equal time to perform. The total time required to velocity profile and discrete depth sample each of the City of Wolfforth PWS wells averaged approximately 36 hours (3 12-h days), including daily setup and breakdown, which required approximately 2 h total per day. Operating the test systems using a 24-h rotating shift labor scheme would reduce overall test time and increase time efficiency, although obviously at the expense of greater personnel requirements.

Discrete depth sampling time efficiency was enhanced by approximately 25% by the addition of vacuum cycles to the procedure. The sampling protocol employed during the City of Wolfforth PWS sampling required approximately 650 mL of sample water to fill two nominal 250-mL sample bottles and to provide additional water for sample container and processing equipment rinsing. Each sampled depth that employed the vacuum required a total of approximately 3 h, including two hose flushing cycles and one sample production cycle, compared with approximately 4 h without the vacuum.

Future System Enhancements

Several future system enhancements could be employed to increase the amount and quality of data obtained and to increase the efficiency of the test systems and procedures. The dye-tracer system injection hose assembly is somewhat elastic, resulting in stretching of the assembly under its own weight as it is lowered to greater depths. The method employed to determine injection depth currently relies on a pinch-roller-style cable length meter installed at the wellhead, which is prone to slipping if the hose is wet. The process of accounting for these potential errors requires an independent measurement of well depth with a separate device (a heavy weight attached to a nonstretching, depth-indexed steel cable) and comparison of the measured depths. Alternative nonstretching hose materials and/or depth measurement methods should be investigated.

The discrete depth sampling procedure requires that the sample hose assembly be flushed at each sampled depth in order to remove remnant water from the previously sampled depth to avoid cross-contamination. During field testing, flushing accounted for approximately 60 to 70% of discrete depth sampling time. During preliminary tests prior to field deployment, it was determined that approximately 100 mL of water remained trapped following a sample retrieval cycle (overpressure expulsion) in the bottom sections of 600-ft-length tube coiled on the hose reel. Following flushing with a 1.0-L volume, two successive samples were obtained from the first depth sampled in Well 8. Constituent analysis results for those two samples were virtually identical within analytical uncertainty, indicating that the presample flush volume was sufficient and may have been more than required. During field testing, presample flush volumes ranged from 1.0 to 1.5 L. Further testing should be conducted to quantify the required flush volumes under different test circumstances.

Additional discrete depth sample hose assemblies of different lengths should be purchased, allowing flexibility in matching hose length with sample depth. Such flexibility would improve sampling efficiency by reducing the flush volume and compressed gas consumption requirements at shallow sample depths. Also, alternative hose assembly construction should be investigated. A hose assembly that employs either multiple pressure hoses or a pressure hose with a larger inside diameter would significantly increase sample volume recovery per cycle, especially under shallow pump submergence conditions.

A preliminary profile of well water parameters using submersible probes to measure conductivity, pH, DO, temperature, etc., might provide information targeting specific depths or depth intervals of interest prior to performing velocity and discrete depth sample profiling. Many probes and configurations are commercially available and should be evaluated for integration into the stratification testing process.

Summary

An aquifer stratification testing system was designed, constructed, and successfully tested. All of the test subsystems functioned properly and either met or exceeded anticipated performance requirements. Velocity profile test results for the Onion Creek

Club well in the Edwards aquifer are consistent with the fracture/solution cavity nature of flow in that aquifer and indicate that most of the flow to the well originates in the bottommost depths of the aquifer. Combined velocity/discharge and discrete depth samples for the two City of Wolfforth PWS wells in the Ogallala indicated variability between flow profiles but consistent aquifer water quality stratification profiles for both wells.

The City of Wolfforth wells tested performance capabilities of the test systems under challenging conditions of low pump discharge rates, small saturated thickness, and shallow sample submergence depths. Results indicate that the system is capable of resolving discharge and water quality profiles over relatively short depth intervals. Future system and procedural modifications should focus on improving and enhancing flexibility and data acquisition capabilities of the system. Future field testing programs should be performed to develop a water quality stratification database to enhance general understanding of groundwater quality and flow processes and provide insight on potential contamination sources.