Ground Water Sampling Bias Observed in Shallow, Conventional Wells

by Stephen R. Hutchins and Steven D. Acree

Abstract

A previous field demonstration project on nitrate-based bioremediation of a fuel-contaminated aquifer used short-screened clustered well points in addition to shallow (10 foot), conventional monitoring wells to monitor the progress of remediation during surface application of recharge. These well systems were placed in the center and at one edge of each of two treatment cells. One cell received recharge amended with nitrate (nitrate cell), and the other received unamended recharge (control cell). Data from the clustered well points were averaged to provide a mean estimate for comparison with the associated conventional monitoring well.

Conservative tracer profiles were similar for each of the four systems, with better fits obtained for well systems located at the edge of the treatment cells. However, aromatic hydrocarbon and electron acceptor profiles varied greatly for the two center well systems, with the conventional monitoring well data suggesting that remediation was proceeding at a much more rapid rate than indicated by the cluster well points. Later tests with an electromagnetic borehole flowmeter demonstrated a significant vertical flow through the wellbore of the conventional monitoring well under simulated operating conditions. This created an artifact during sampling, thought to arise from preferential flow of recharge water from the water table to deeper portions of the contaminated zone resulting in several effects, including an actual decreased residence time of water sampled by the conventional well. These data provide additional evidence that conventional monitoring wells may be inadequate for monitoring remediation in the presence of significant vertical hydraulic gradients, even for fairly shallow homogeneous aquifers.

Introduction

An understanding of ground water chemistry and hydrologic processes is generally gained through the use of conventional monitoring wells. In fact, these wells are often used to provide information on the rate and extent of remediation of contaminated aquifers (National Research Council 1994). It is generally recognized that conventional monitoring wells can provide only a "composite" picture of the subsurface environments contacted by the entire screened interval, and the extent to which individual transmissive layers influence the overall sample quality depends not only on the physical characteristics of both the well and the subsurface environment, but also on the methods by which representative samples are obtained (Robbins and Martin-Hayden 1991; Martin-Hayden et al. 1991; Puls and Paul 1997). The use of vertical well points or cluster wells can provide much greater definition of subsurface heterogeneities and can isolate areas that, although in close proximity, may be quite different with respect to chemical and microbiological properties (Smith et al. 1991; Martin-Hayden and Robbins 1997). The disadvantage of this approach is that individual subsurface layers that contribute to the bulk ground water flow may be missed, and a larger number of samples are therefore required to define the overall subsurface environment. Conventional monitoring wells are therefore still used extensively for assessing remediation, although in certain instances modifications are made to sampling techniques (e.g., multilevel packers) to allow depth-discrete sampling (Powell and Puls 1993; Puls and Paul 1997). However,

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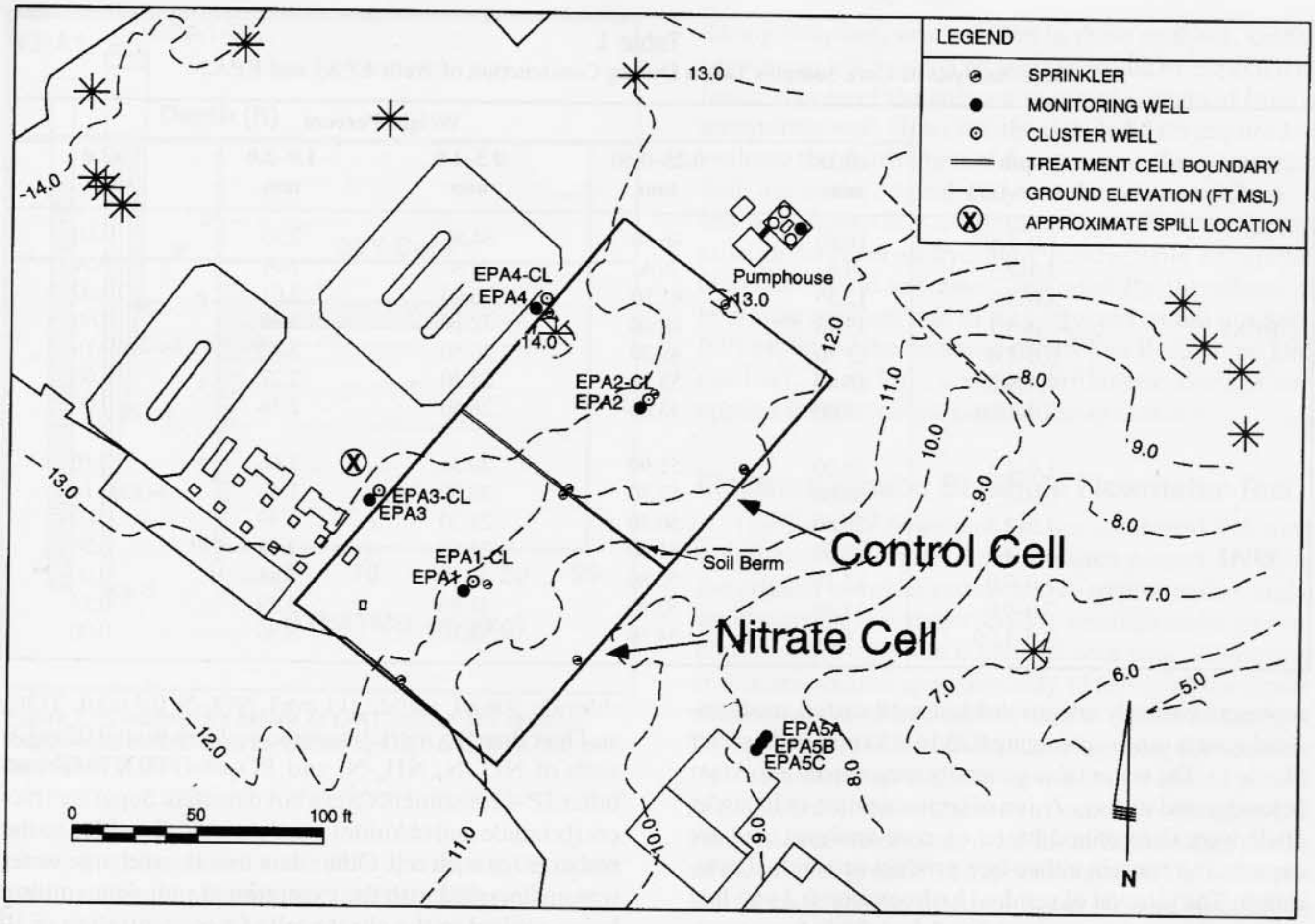

Figure 1. Site map of pilot demonstration project showing locations of pumphouse, sprinklers, monitoring wells, and cluster wells. The approximate spill location is also shown.

another problem with conventional monitoring wells has been identified: the potential for redistribution of contaminants due to vertical flow within the wellbore (Reilly and LeBlanc 1998). This was observed for a long-screened (40 foot) well that had significant physical and chemical heterogeneities in the aquifer adjacent to the screen. Although the potential for vertical flow would be evident in this system due to the possible existence of vertical hydraulic gradients, the same problem has been observed for long-screened wells in relatively homogeneous aquifers (Church and Granato 1996).

Anomalies in water-quality indices between short-screened cluster wells and conventional monitoring wells had also been observed (Hutchins 1998) during a previous field demonstration project on bioremediation of a fuel-contaminated aquifer using sprinkler recharge (Hutchins et al. 1998). These indices included conservative tracers (bromide and chloride), added or indigenous electron acceptors (oxygen, nitrate, and sulfate), and BTEXTMB (benzene, toluene, ethylbenzene, xylenes, and trimethylbenzenes). It was hypothesized that vertical flow within the wellbore was partially responsible for these discrepancies, even though the shallow conventional monitoring wells were screened only in the upper 6 to 9 foot sections of the aquifer. The recharge conditions were therefore simulated during a subsequent field trip and an electromagnetic borehole flowmeter was used to estimate the impact of vertical wellbore flow during ambient (nonpumping) conditions. This information was used to determine whether vertical wellbore flow occurred during the field demonstration project, and how it could lead to an overestimate of the extent of bioremediation if data were obtained solely from shallow, conventional monitoring wells.

Experimental Design

Field Demonstration Project

The field site is located at Eglin Air Force Base, Florida, where a shallow aquifer is contaminated with JP-4 jet fuel (Figure 1). Extensive information on site characterization has been published elsewhere (R.F. Weston Inc. 1984; EA Engineering 1987; Thomas et al. 1995; Sweed et al. 1996). In brief, the terrain is relatively flat, with the subsurface consisting of a relatively homogeneous 30- to 40-foot-thick shallow sand-and-gravel aquifer that extends down to contact the Pensacola Clay confining unit. Ground water flows to the southeast with an average linear velocity of 500 ft/yr, and interpretation of cone penetrometer logs suggests that the hydraulic conductivity distribution of the sand is relatively homogeneous and ranges from approximately 30 to 140 ft/day (Sweed et al. 1996). The estimated porosity is 35% to 45%. Aquifer sediments are texturally mature to submature quartz sands commonly associated with a beach environment, and core samples in the area of contamination

Table 1
Sieve Analysis of Core Samples Taken During Construction of Wells EPA1 and EPA2

Well Location	Depth (ft from GS)	Weight Percent <0.25 mm	0.25–0.50 mm	0.5–1.0 mm	1.0–2.0 mm	>2.0 mm
EPA1	1.0–1.5	16.60	46.50	34.50	2.33	0.10
	2.0–2.5	NA	NA	NA	NA	NA
	3.0–3.5	12.50	41.50	26.40	3.01	16.60
	4.0–4.5	15.40	48.60	32.10	3.89	0.00
	5.0–7.0	21.70	49.30	26.50	2.40	0.18
	7.0–9.0	16.40	53.10	28.20	2.27	0.00
	9.0–11.0	18.00	53.00	26.60	2.36	0.00
EPA2	1.0–1.5	26.00	51.90	20.30	1.66	0.16
	2.0–2.5	25.60	52.30	20.50	1.48	0.14
	3.0–3.5	26.80	50.30	21.30	1.49	0.00
	4.0–4.5	22.80	51.60	23.10	2.16	0.36
	5.0–7.0	13.10	52.80	30.30	3.84	0.00
	7.0–9.0	11.40	52.90	31.80	3.79	0.14
	9.0–11.0	9.41	54.50	32.10	3.96	0.00

represent basically unconsolidated, well-sorted, medium-sized quartz sands, averaging 0.25 to 0.50 mm in diameter (Table 1). The water table generally ranges from 3 to 5 feet below ground surface. Zones of contamination (>10 mg/kg JP-4) were determined based on core analyses, and are expected to contain either free product or residual saturation. The interval of residual hydrocarbon is 4 to 5 feet thick in the source area and 2 to 3 feet thick downgradient. The bottom of the contaminated zone (<10 mg/kg JP-4) ranges from 4 to 7 feet below land surface. Hence, most of the JP-4 is located below the water table.

Details of the field demonstration project on nitrate-based bioremediation have been described by Hutchins et al. (1998). In brief, the objective of the demonstration project was to compare the extent of remediation using sprinkler recharge with and without nitrate addition as a supplemental electron acceptor. Two 100 foot by 100 foot treatment cells were delineated for treatment, with the southwest cell being designated the nitrate cell and the northeast cell designated the control cell (Figure 1). The land surface of the cells was generally covered with bermuda grass, although vegetation was more sparse adjacent to the source area. Plastic sheeting was installed in a trench separating the two treatment cells to minimize crossover during infiltration to the water table. The trench depth ranged from 2.0 to 2.5 feet (at the water table) on the east side to 4.0 to 4.5 feet (above the water table) on the west side. A soil berm was then built over the filled trench to prevent runoff onto the nitrate cell, since the land surface sloped down toward the southeast. Other than this, there was no surface or subsurface construction for hydraulic containment. The recharge water was obtained from the Floridan Aquifer, the same source that provided ground water for that part of the base. The water was clean, with approximately 15 mg/L sodium, 3 mg/L potassium, 25 mg/L calcium, 15 mg/L magnesium, and less than 0.05 mg/L iron and manganese. The pH was 7.6 and there was no measurable dissolved oxygen. The recharge water contained approximately 7 mg/L chloride, 9 mg/L sulfate, 0.1 mg/L NO_3-N, 0.3 mg/L TOC, and less than 0.5 mg/L bromide, and less than 0.05 mg/L each of NO_2-N, NH_4-N, and PO_4-P. BTEXTMB and other JP-4 constituents were not detected. Separate tracers (bromide and chloride) were periodically added to the recharge for each cell. Other than this, the recharge water was unamended with the exception of potassium nitrate being applied to the nitrate cell at a concentration of 10 mg/L NO_3-N. Recharge was applied continuously to each cell at a rate of 11.0 to 11.5 gal/min (gpm) per cell, corresponding to about 2.5 inches per day (in/d).

Application water and ground water quality were monitored continuously during system operation using both conventional and clustered monitoring wells. For each cell, a conventional well and a cluster of short-screened well points were placed in the center and at one of the edges (Figure 1). The conventional wells were constructed of 2 inch Schedule 40 PVC and screened 1 to 11 feet below ground surface. The clustered wells consisted of five individual wells per cluster and were installed separately, adjacent to the fully penetrating wells, using a Geoprobe®. Each cluster well was constructed of ¼-inch polypropylene tubing with a 2.5-inch 80-mesh steel screen. The top of each cluster well was sealed with a Teflon® plug valve. The wells were installed 4.0, 5.0, 6.5, 8.5, and 11 feet below ground surface for each cluster location (Figure 2). For sampling the 2 inch wells, a Grundfos™ submersible pump was used. The pump was set sequentially at the top, bottom, and middle of the water column and pumped for five minutes at 0.79 gpm for each level to purge each well. This resulted in the removal of approximately 10 well volumes. The flow rate was then reduced to approximately 0.13 gpm, and samples were obtained from the middle of the water column. For sampling the small cluster wells, a peristaltic pump with multiple heads was connected directly to the wells. The cluster wells were all purged and sampled simultaneously. The wells were purged for five minutes at 0.026 gpm (>10 well volumes) and sampled at the same rate. Once sampling was com-

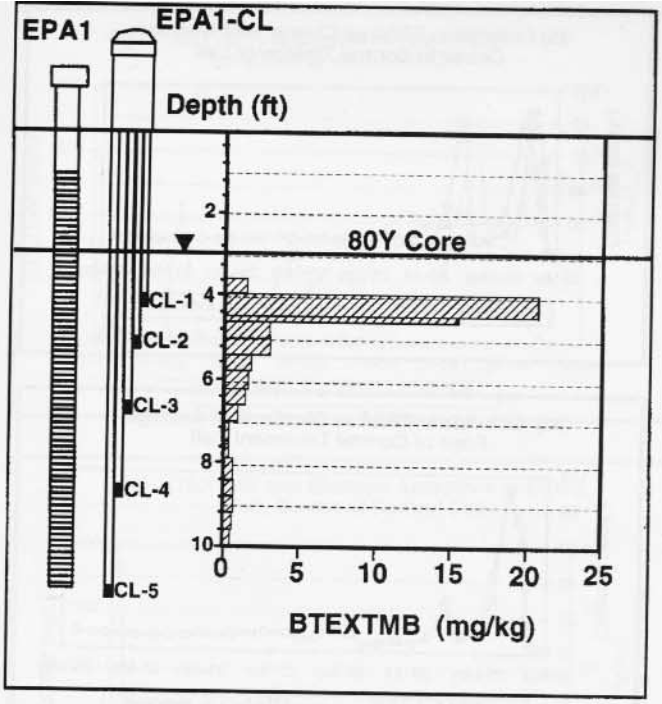

Figure 2. Construction details of EPA1 cluster well showing locations of well screens in relation to EPA1 and BTEXTMB contamination in 80Y core.

plete, the plug valves were closed, trapping the water column and not allowing air to re-enter the well lines.

Water Quality Indices and Data Analysis

Although several water-quality parameters were measured during the pilot demonstration project, this report focuses on the following groups: (1) conservative tracers (bromide and chloride); (2) added or indigenous electron acceptors (oxygen, nitrate, and sulfate); and (3) dissolved contaminants (BTEXTMB). Details on sample handling and analytical procedures have been described by Hutchins et al. (1998). Short-term tracer studies were conducted at two different times: (1) at the start of operation, to evaluate water movement when the vadose zone was initially low in water content; and (2) during operation, to evaluate water movement under saturated operating conditions. Two tracers, bromide for the nitrate cell and chloride for the control cell, were used to differentiate between the recharge for each cell. Samples were collected more frequently from the cluster wells during these studies than at other times. For all other parameters, including the conservative tracers outside of these time periods, samples were collected from both cluster wells and conventional wells at two-week intervals. To provide a comparison to the conventional well data, a simple average of cluster well concentrations was calculated from each of the five clustered well points for each parameter. This was not always the case for EPA4, located at the edge of the control cell (Figure 1), since the upper well point of this cluster was often above the water table. Note that a flow-weighted average, whereby constituent concentrations in samples from each cluster well are weighted as a function of the relative influx of water from each interval into the conventional monitoring well during sampling, was not used in these analyses. Under certain conditions, such an average would be expected to better represent the composite sample obtained from a monitoring well. However, the detailed data required to evaluate the distribution of flow entering the monitoring wells during the original study were not obtained. Regardless, a calculated flow distribution based solely on the relative differences in hydraulic conductivity of aquifer materials would not have accounted for the effects of hydraulic gradient due to recharge and would not have fully explained the observed results (see Results and Discussion). Therefore, a simple arithmetic average was applied for the comparisons of this evaluation.

Electromagnetic Borehole Flowmeter Test

The borehole flowmeter test was conducted following completion of the pilot demonstration project. By then, the original sprinkler system had been dismantled, and a smaller system was constructed to simulate recharge conditions. Each cell was equipped with four 360-degree sprinklers, located approximately 13 feet from the center monitoring well. Each set was used to deliver recharge at 11.5 gpm to provide a circular coverage of approximately 74 feet in diameter with the monitoring wells in the center. Because of the decreased surface area coverage, this resulted in approximately twice the infiltration rate as that of the original system. The sprinkler system for the nitrate cell (EPA1) was operated for about 16 days, resulting in a water table mound of approximately 1 foot which developed in two to three days (data not shown). This compared favorably with the water table mound of 0.8 foot observed in the previous pilot demonstration project (Sweed et al. 1996).

The electromagnetic borehole flowmeter survey used a 1.0 inch I.D. probe. This probe is capable of measuring flow rates as low as approximately 0.03 gpm in the laboratory, which corresponds to a linear velocity of approximately 20 cm/min through the probe. The probe design is based on Faraday's law, which states that the voltage induced by a conductor moving through a magnetic field is directly proportional to the velocity of the conductor. The major components of the probe include an electromagnet and a pair of electrodes mounted at right angles to the poles of the magnet. The voltage generated as the conductive fluid (ground water) flows through the induced magnetic field is proportional to the average water velocity through the probe (Molz and Young 1993). Prior to conducting the study, the flowmeter was calibrated in a test cell constructed of materials identical to those of the study wells (i.e., 2 inch Schedule 40 PVC). Calibration was accomplished by using the borehole flowmeter to measure water flow in the casing at known flow rates chosen to span the range of flow rates (approximately 0.037 to 2.4 gpm) anticipated to be observed in the field. This allowed volumetric flow rates in the test cell to be correlated with flowmeter voltage measurements induced by water velocity. Data obtained during the calibration phase indicated a linear meter response over the range of potentially applicable flow rates. Testing was conducted on EPA1 using the methods of Molz et al. (1994), with ambi-

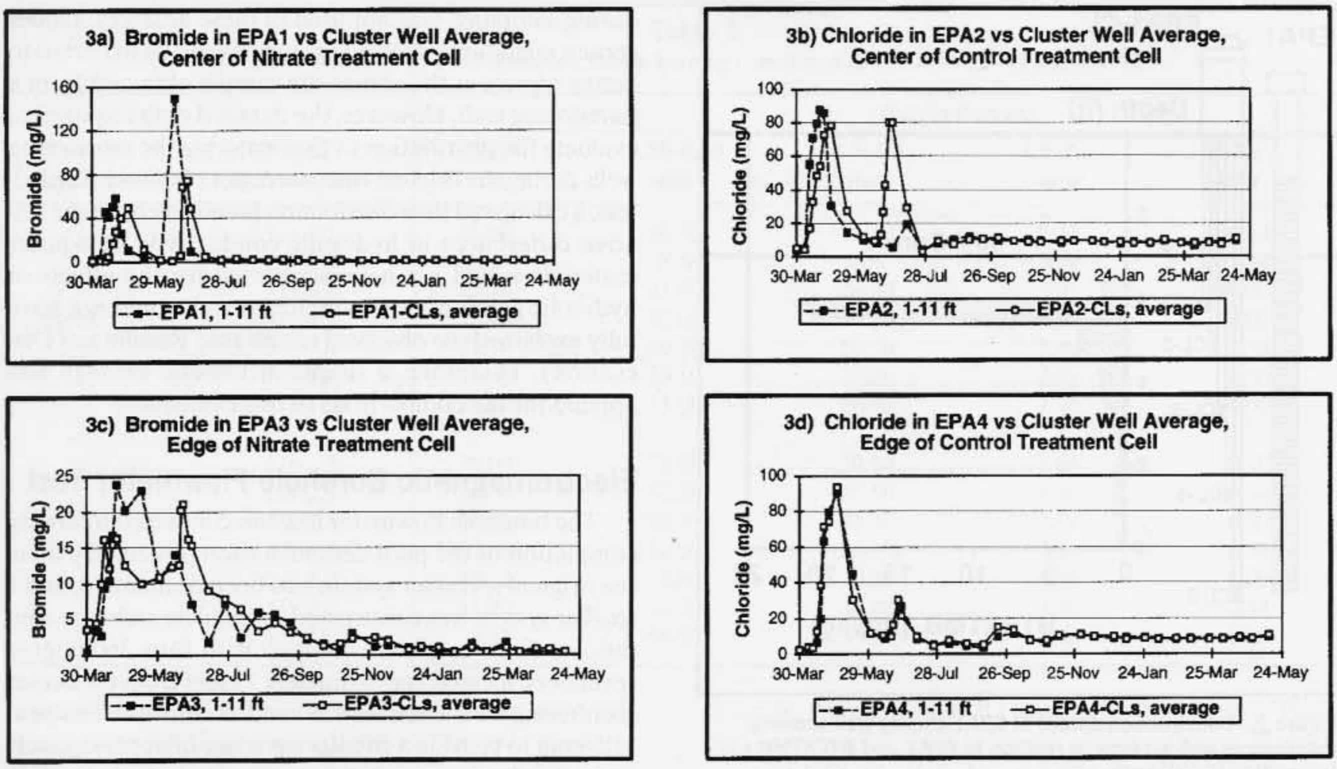

Figure 3. Comparison of tracer data in conventional monitoring wells vs. cluster well averages for (a) EPA1, (b) EPA2, (c) EPA3, and (d) EPA4.

ent vertical flow rates being measured from total depth to the top of the saturated well screen using the 1.0 inch I.D. probe at 1 foot intervals. Measurements under ambient conditions provided information regarding the vertical component of the hydraulic gradient and quantification of water flow entering and exiting the well from chosen intervals at the time of the test.

Results and Discussion

Comparison of tracer data between the conventional monitoring wells and the cluster well averages generally provided a reasonable match with respect to time and concentration, even though time shifts were evident for the two center wells (Figure 3). Time shifts were more apparent in the center wells, especially during the initial tracer study, when breakthrough occurred more rapidly within the conventional wells compared to the cluster well averages (Figures 3a and 3b). Other than these time shifts, the overall tracer profiles were similar. In contrast, concentrations of nonconservative parameters such as BTEXTMB, dissolved oxygen, nitrate, and sulfate were often different, especially in the center wells (Figure 4). For example, the conventional well in the nitrate cell showed a rapid loss of BTEXTMB and a significant breakthrough of nitrate and, to a lesser extent, oxygen (Figure 4a). However, the cluster wells indicated that the ground water still had high concentrations of BTEXTMB, and nitrate and oxygen levels were much lower (Figure 4b). Hence, the rate and extent of remediation was overestimated with the conventional monitoring well. Similar discrepancies in the results, although to a lesser scale, were obtained for the well pairs in the center of the control cell (Figures 4c and 4d). Nitrate was not added to the recharge for this cell; however, sulfate concentrations were approximately 10 mg/L in the unamended recharge, and other data indicated that iron and sulfate were the primary electron acceptors for this cell (Hutchins et al. 1998). Although the differences were not as apparent as with the nitrate cell, the conventional monitoring well data also indicated a greater degree of electron acceptor breakthrough and BTEXTMB removal than the cluster well data.

In contrast, comparative profiles for electron acceptors and BTEXTMB were more variable in the well pairs at the edges of the treatment cells (Figure 5). For both treatment cell edge pairs, breakthroughs of electron acceptors were similar. However, for the nitrate cell, BTEXTMB removal was highly variable based on the conventional monitoring well data whereas a gradual decline was observed based upon the cluster wells (Figures 5a and 5b). The sharp decreases in BTEXTMB concentrations observed in the conventional well generally corresponded to periods of rainfall events and the subsequent increases in the elevation of the water table (data not shown). For the edge well pair of the control cell, this BTEXTMB concentration discrepancy was not as apparent (Figures 5c and 5d). Because this well pair was adjacent to a large concrete pad (Figure 1) and had been covered with clay fill, it is possible that rainfall events had less of an impact.

The electromagnetic borehole flowmeter was used to characterize ground water flow in conventional well EPA1 during simulated recharge conditions. Under these intensive recharge conditions, ground water flow within EPA1 was downward (Figure 6a) due to the vertical gra-

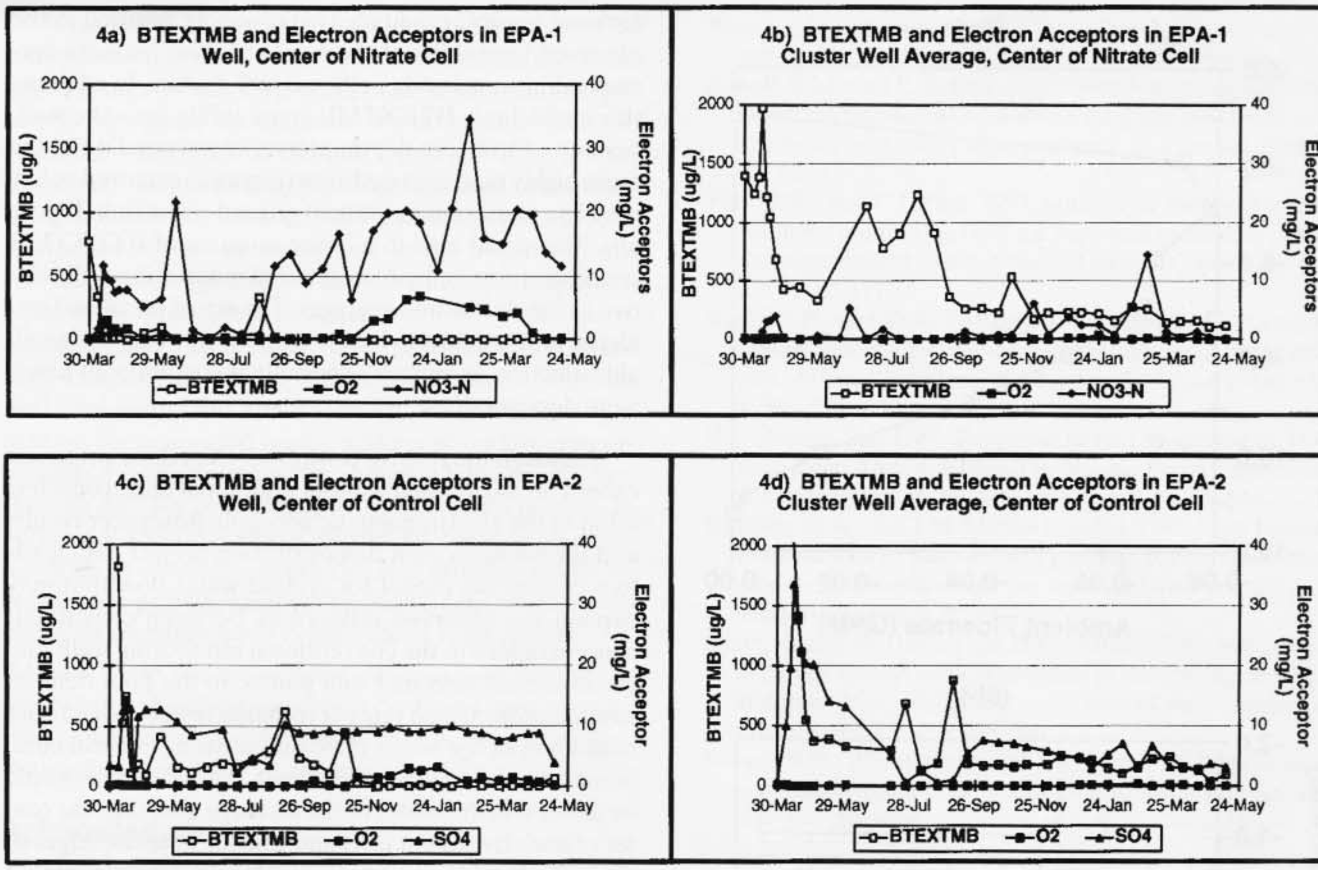

Figure 4. BTEXTMB and electron acceptor levels in (a) EPA1 monitoring well, (b) EPA1 cluster wells, (c) EPA2 monitoring well, and (d) EPA2 cluster wells.

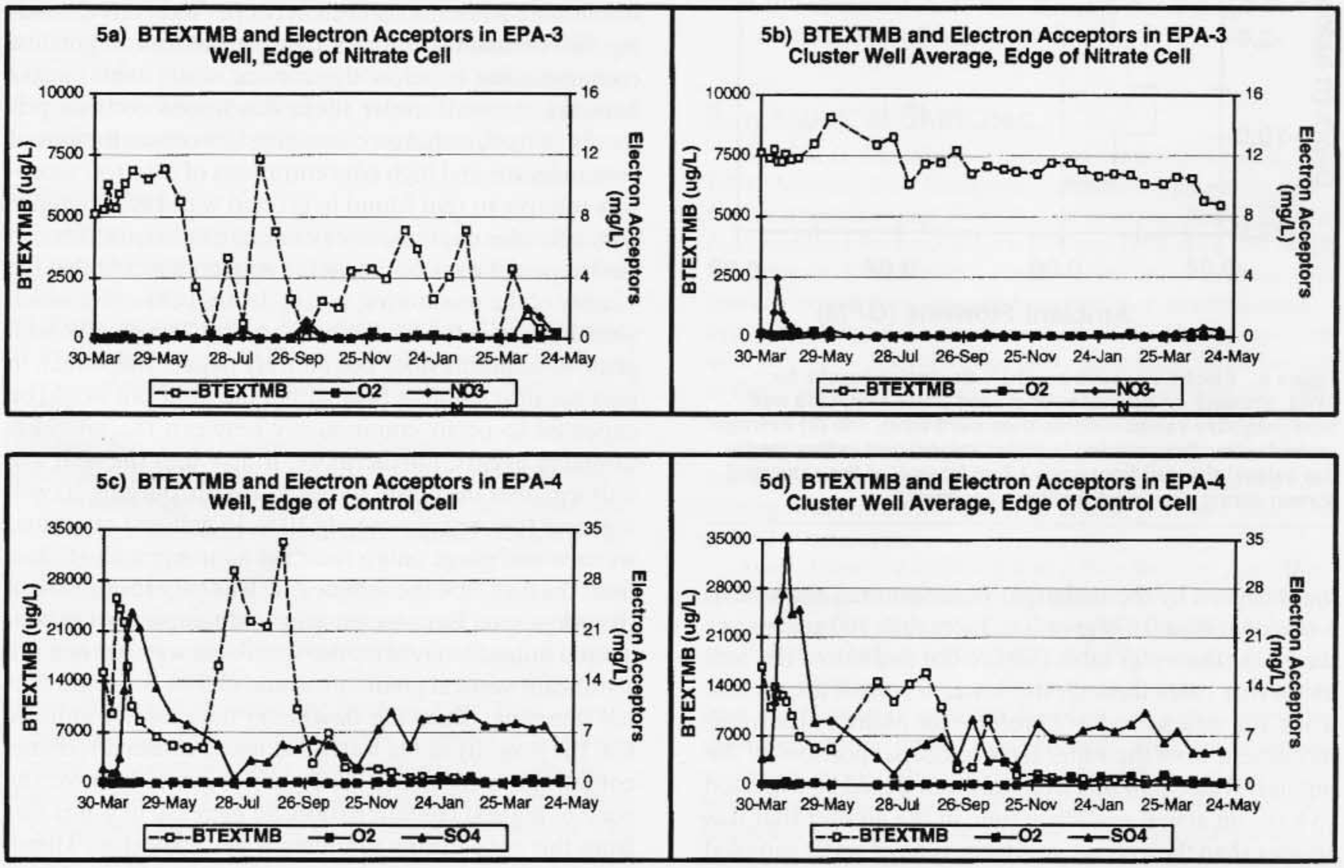

Figure 5. BTEXTMB and electron acceptor levels in (a) EPA3 monitoring well, (b) EPA3 cluster wells, (c) EPA4 monitoring well, and (d) EPA4 cluster wells.

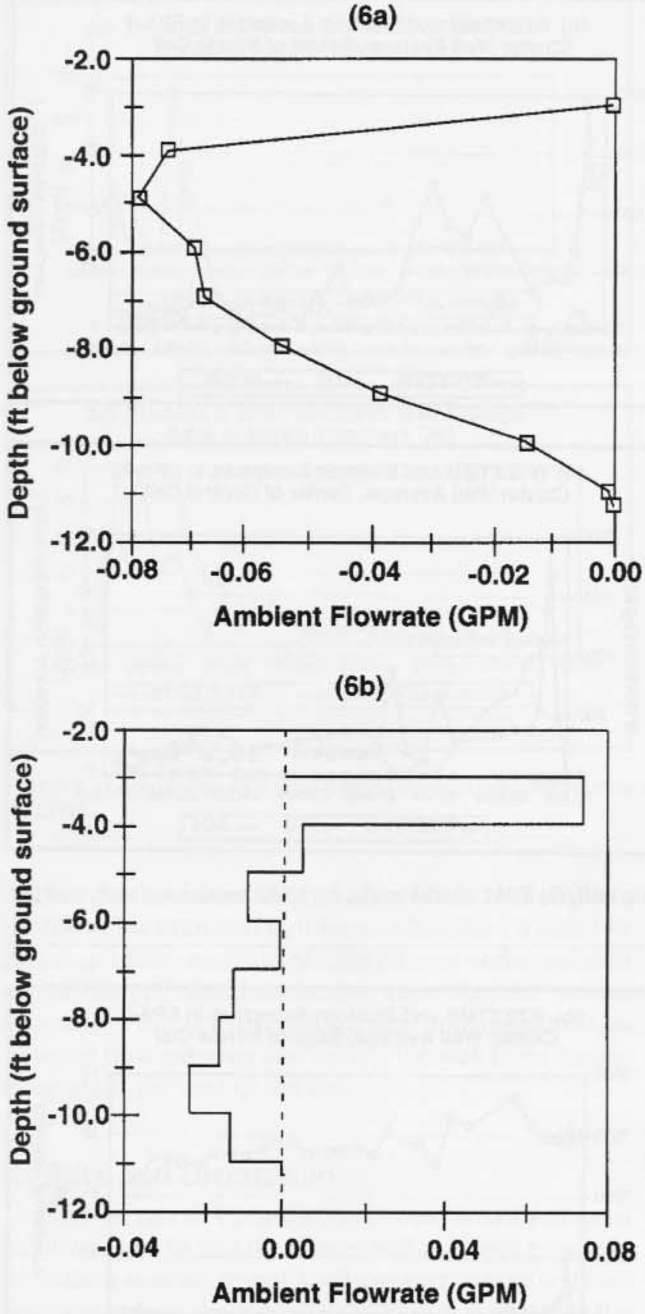

Figure 6. Electromagnetic borehole flowmeter results for EPA1, showing (a) flow rates of ground water within the wellbore (negative values indicate downward flow), and (b) average ground water flux entering (positive values) and exiting (negative values) the well from each 1 foot interval of the saturated screen during simulated recharge conditions.

dient caused by the recharge. Water entered the well at a rate of at least 0.079 gpm (i.e., more than 100 gallons per day) near the water table (Figure 6b) and exited the well at varying rates through the lower 6 feet of the screen. Thus, the well served as a preferential pathway for water movement from the water table to deeper portions of the aquifer. Water sampled from this well would be expected to have an actual residence time in the aquifer that was shorter than the average residence time of water sampled by the clustered well points. A significant fraction of the water obtained during sampling may also have come directly from the interval near the water table due to the ambient vertical gradient. This probably resulted in the observed time shifts of the tracer profiles between corresponding monitoring wells and well clusters. In addition, this could limit BTEXTMB mass influx into the well, since the 4 to 6 foot depth interval corresponding to the more highly contaminated area (Figure 2) occurred below the zone of maximum influent ground water flux (Figure 6b). This could lead to a lower estimate of BTEXTMB concentrations in the conventional monitoring well relative to the cluster well averages. The actual decreased residence time would also lead to a slower rate of use of available electron acceptors, especially if contaminant levels were decreased.

Although the relative contributions of these processes cannot be ascertained without additional data, consideration of the electromagnetic borehole flowmeter results and the previous pilot demonstration project data leads to a conceptual model for ground water flow that may explain the observed differences between constituent concentrations in the conventional monitoring wells and the discretely screened well points. In the pilot demonstration project, high rates of recharge result in significant mounding of the water table and a strong vertical component to the hydraulic gradient. These effects would be greatest in the center of the recharge area (i.e., the center of each treatment cell) and smaller near the edges of the cells. Strong vertical gradients result in an ambient flow field within conventional wells screened across the water table, which act as open conduits connecting zones of different hydraulic head. Water enters the well near the water table and exits the well in deeper intervals (including the contaminated zone). Because the zone of greatest contamination is below the current water table, water entering the well under these conditions consists primarily of fresh recharge, containing low concentrations of contaminants and high concentrations of electron acceptors relative to that found in ground water from deeper zones. Under continuous operating conditions, this can lead to partial remediation of the aquifer matrix within the vicinity of the monitoring well at depths below that which would be expected for the bulk aquifer. Purging the well prior to sampling does not entirely negate this effect, in part because ambient flow within the wellbore would be expected to occur continuously between the biweekly sampling events. Furthermore, if flow into the well was uniform over the entire screened length, purging 10 well volumes (i.e., approximately 10 to 16 gallons) of ground water would purge only a radius of approximately 0.5 feet from the well, and the sample rate was only about 16% of the purge rate. Because purging and sampling were conducted immediately after the sprinklers were turned off, significant vertical gradients would still be expected during sampling. Since the flowmeter data clearly indicate that the majority of the water flux under constant recharge conditions is from the water table, located above the zone of highest contamination, an increase in water flux from this zone during sampling is also expected. Therefore, it is not surprising that water samples obtained from the conventional monitoring well are less contaminated than those obtained from the corresponding cluster wells.

Regardless of the exact mechanism, the extent of bioremediation can be clearly overestimated using the conventional monitoring wells, even in this shallow, relatively homogeneous aquifer. It should be noted that the recharge rate used in the electromagnetic borehole flowmeter study was roughly twice that used in the original pilot demonstration project, and so the vertical flow components were probably less in the original study. However, the magnitude of the vertical flow component is not as an important as its effect over time in this system, especially under continuous recharge conditions. Vertical flow components can be expected for other systems as well, including land application of waste water, irrigation of contaminated areas, and remediation systems relying on surface application, drainage lines, or injection through partially penetrating wells. These data provide an additional example where cluster well points or other methods of obtaining depth-discrete samples would be required to provide an accurate assessment of either the extent of contamination or the rate of remediation.

Editor's Note: The use of brand names in peer-reviewed papers is for identification purposes only and does not constitute endorsement by the authors, their employers, or the National Ground Water Association.

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