Implications of Observed and Simulated Ambient Flow in Monitoring Wells

by Alper Elci[1], Fred J. Molz III[2], and William R. Waldrop[3]

Abstract

A recent paper by Hutchins and Acree (2000) has called attention to ground water sampling bias due to ambient (natural gradient-induced) flows in monitoring wells. Data collected with borehole flowmeters have shown that such ambient flows are ubiquitous in both confined and unconfined aquifers. Developed herein is a detailed three-dimensional model of flow and transport in the vicinity of a fully penetrating monitoring well. The model was used to simulate a measured ambient flow distribution around a test well in a heterogeneous aquifer at the Savannah River Site (SRS) near Aiken, South Carolina. Simulated ambient flows agreed well with measurements. Natural flow was upward, so water entered the well mainly through high K layers in the lower portion of the aquifer and exited through similar layers in the upper portion. The maximum upward discharge in the well was about 0.28 L/min, which implied an induced exchange of 12 m^3/month from the bottom half of the aquifer to the upper half. Tracer transport simulations then illustrated how a contaminant located initially in a lower portion of the aquifer was continuously transported into the upper portion and diluted throughout the entire well by in-flowing water. Even after full purging or micropurging, samples from such a well will yield misleading and ambiguous data concerning solute concentrations, location of a contaminant source, and plume geometry. For all of these reasons, use of long-screened monitoring wells should be phased out, unless an appropriate multilevel sampling device prevents vertical flow.

Background

Conventional monitoring wells are often used to obtain information about ground water chemistry and plume geometry. The gathering of information is accomplished by collecting ground water samples for determination of the distribution and magnitude of contamination level and for monitoring the progress of remedial actions that are ongoing or have been taken previously. In all of these applications, the location and concentration of the ground water sample is critical to a realistic interpretation of contaminant transport and fate in an aquifer. What may not be widely realized is that the installation of the monitoring well itself may set up a local vertical flow system due to a natural vertical hydraulic gradient at the well location. The well then acts as a "short circuit" along this gradient, with the resulting flow in the wellbore (ambient flow) often of sufficient magnitude to compromise the integrity of any samples collected from the well. Such an effect has been reported in a few previous studies (Reilly et al. 1989; Church and Granato 1996; Hutchins and Acree 2000).

As summarized in Table 1, application of sensitive borehole flowmeters, has enabled ambient flow to be documented in several past studies (Molz and Young 1993; Molz et al. 1994; Church and Granato 1996; Boman et al. 1997; Hutchins and Acree 2000; Crisman et al. 2000). For most of the wells listed in Table 1, whether the well screen penetrated the aquifer fully was not documented. The last five wells shown in Table 1 were selected from a group of flowmeter tests performed on a total of 142 wells at 16 sites in 12 states. Flow was measured with an electromagnetic flowmeter, and in 73% of the cases a measurable amount of ambient flow was observed. The majority of the ambient flow cases, 62%, displayed a downward ambient flow, 31% of the cases displayed upward flow, and in 7% of the cases a mixed type of ambient flow (upward and downward) was observed. However, it should be noted that such percentages may vary widely between sites. Flow directions, patterns, and magnitudes are different in each case study, and it is difficult to determine ambient flow direction and magnitude without a direct measurement. In other words, no generalization can be made for the possible maximum flow rate in a wellbore for a confined aquifer. This situation can be explained by the fact that the media are heterogeneous in various degrees and that the hydrogeological characteristics are different from site to site. Even flowmeter measurements in different wells at the same site often reveal different patterns and magnitudes of ambient wellbore flow (Boman et al. 1997; Molz et al. 1994; Church and Granato 1996) due to spatial and temporal changes of conditions in the aquifer.

Only a few previous studies of possible sampling bias due to ambient flow in monitoring wells have been published. Reilly et al. (1989) studied a hypothetical ground water system numerically in order to demonstrate that significant amounts of flow can occur within long-screen wells installed in homogeneous aquifers. A

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Received September 2000, accepted April 2001.

Reference	Location and Type of Aquifer, Type of Medium	Direction of Ambient Flow	Screen Length (m)	Max. Ambient Flow (L/min)
Boman et al. (1997)	Louisiana BAP1[2]; clay, silt and sand mixtures	downward	6.71	0.015
	Louisiana BAP2	downward	6.1	0.031
	Aiken, SC; confined; K_{avg} = 5.7 m/day	upward	12.2	0.26
Molz et al. (1994)	Mobile, AL; confined; fluvial sediments, medium sand with silt and clay fines	upward	21.5	3.1–3.6
	Mobile, AL[2] (deeper well); confined	mixed	20	1.75
Hutchins and Acree (2000)	Eglin AFB, FL[2]; unconfined; sand and gravel, K = 9.1–42.7 m/day	downward	3.1	0.30
Church and Granato (1996)	Massachusetts; unconfined; sandy; K = 1–15 m/day	upward	18	0.48
	Massachusetts; unconfined; sandy; K = 1–50 m/day	mixed	21	0.04
Crisman et al. (2000)	Aiken, SC.; unconfined; sand and clay; K_{avg} = 0.35 m/day	upward	4.2	0.34
Waldrop (unpublished)	Coastal Virginia, confined	downward	5.8	0.276
	Central Ohio, unconfined	upward	12.6	6.22
	Western Texas	downward	9.1	2.3
	Idaho, unconfined	upward	56.6	1.03
	Central Louisiana	mixed	21.3	0.57

Table 1
Summary of Documented Ambient Wellbore Flow[1]

[1]For most of the wells, whether the well screen penetrated the aquifer fully was not documented. The five wells measured by Waldrop are a selected group out of 142 wells at 16 sites.
[2]Known fully penetrating monitoring wells.

numerical simulation was performed on an unconfined aquifer system assuming two-dimensional regional flow. The authors applied a vertical hydraulic gradient to the regional model by assigning an areal recharge rate at the free surface of the aquifer. A local system that consisted of a three-dimensional section of the regional model was simulated in order to analyze and quantify the wellbore flow. In the center of the local system, one gridblock was assigned a much higher hydraulic conductivity than the surrounding aquifer. This gridblock represented an 18.3 m long well screen that fully penetrated the aquifer. The most important result of the study by Reilly et al. (1989) was the clear prediction that a flow rate detectable with borehole flowmeters will occur in aquifers with very small typical vertical head differences. It was also found that inflow to the wellbore was highly concentrated near the top of the well screen and that outflow was concentrated near the bottom. The authors concluded that samples from contaminant monitoring wells with long screens might be almost useless for quantifying the concentration of contaminants. Their warning, however, seems to have been largely ignored, possibly because the mixing and dilution process was not simulated in detail and compared with field data. Over the years, however, there has been a gradual shift away from long-screened monitoring wells toward short-screened cluster wells or multilevel samplers.

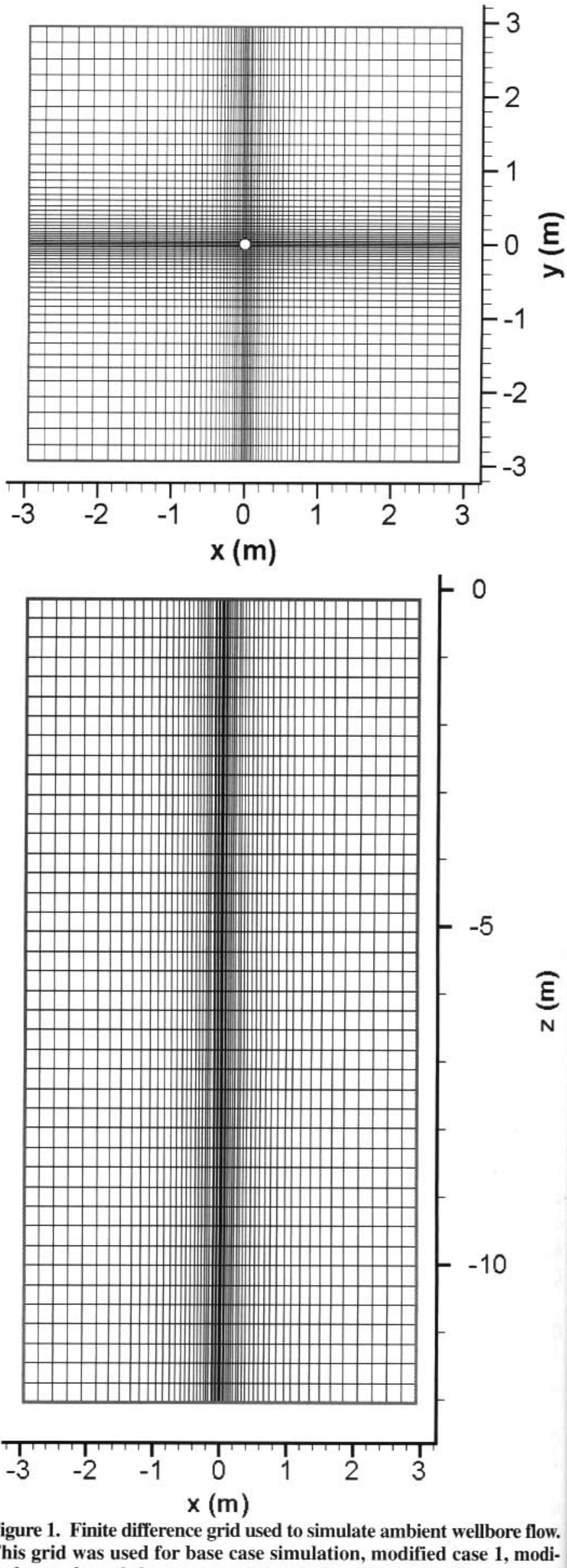


Figure 1. Finite difference grid used to simulate ambient wellbore flow. This grid was used for base case simulation, modified case 1, modified case 2, and the transport simulation. The grids for modified case 3 had 10 and 20 layers; the horizontal dimensions were the same. (a) plan view; (b) side view.

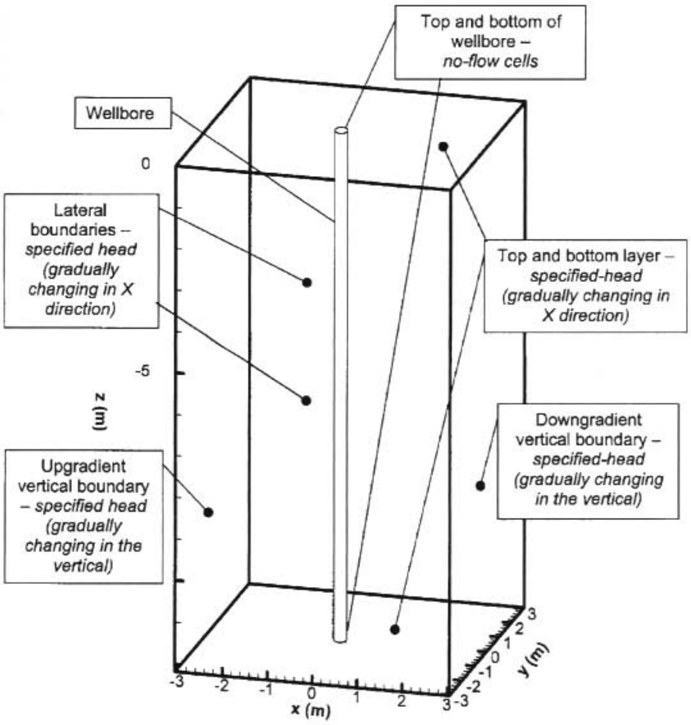


Figure 2. Diagram showing boundary conditions of the model.

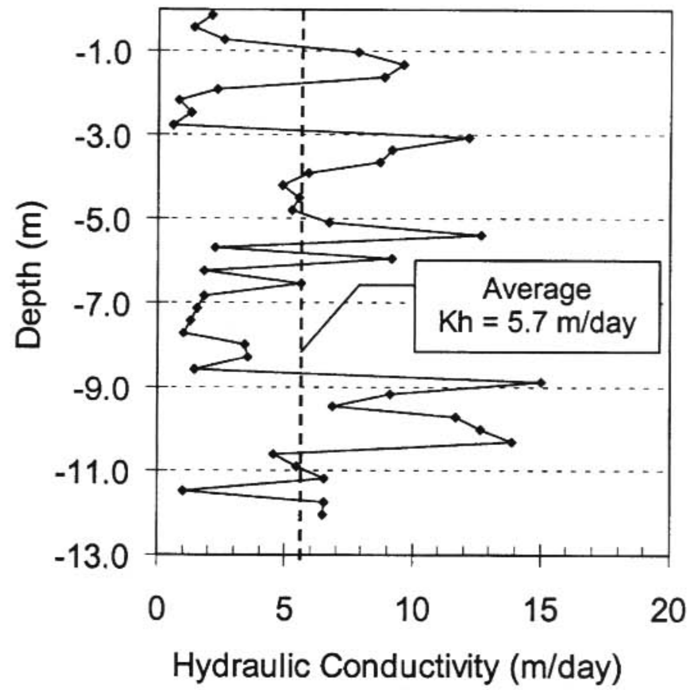


Figure 3. Hydraulic conductivity distribution by flowmeter analysis for well P26-M1 (Boman et al. 1997).

A more recent study by Hutchins and Acree (2000) demonstrated the bias of ground water sampling observed in long-screened (conventional) monitoring wells. Short-screened clustered well points were used in addition to conventional monitoring wells to observe the progress of a nitrate-based bioremediation of a shallow, fuel-contaminated, aquifer. Ground water quality data from the clustered short-screen wells were averaged to provide a mean estimate for comparison with the associated conventional monitoring well. Tests with an electromagnetic borehole flowmeter demonstrated a significant upward ambient flow (0.30 L/min, see Table 1) through the wellbore of the conventional monitoring well. The authors found experimentally that the extent of bioremediation was clearly overestimated using the conventional monitoring wells due to dilution of the samples caused by ambient flow.

Church and Granato (1996) conducted a field experiment comparing water-quality constituents, specific conductance, geophysical measurements, and wellbore hydraulics in long-screen wells and adjacent vertical clusters of short-screen wells to show bias in ground water data due to ambient flow. Their study indicated that sampling from a long-screen monitoring well in ambient flow would be either diluted or concentrated, depending on the vertical head distribution and the actual contaminant plume location.

Purpose and Scope

The purpose of this paper is to present the development and results of a detailed three-dimensional simulation model of flow and transport in the vicinity of a fully penetrating monitoring well in a confined aquifer. There were two types of simulations: flow simulations and tracer transport simulations. Flow simulations were run for four cases, and the tracer transport simulations were performed for two cases. The flow simulation results were needed to determine theoretically the ambient flow distribution, and the magnitude and direction of ambient flow. The first flow simulation, which is named the base case, was performed using hydrogeologic properties in the vicinity of a test well in a confined, heterogeneous aquifer at the Savannah River Site near Aiken, South Carolina. Borehole flowmeter measurements for two wells at that site were already available (Boman et al. 1997). The horizontal hydraulic conductivity distribution calculated from borehole flowmeter data and the associated hydraulic gradient for one of these wells (P26-M1) were used for the base case simulation (Figure 3). The goal of the base case simulation was to predict the measured ambient flow distribution around well P26-M1. The base case simulation results were then compared to field measurements to verify the conceptualization of the wellbore flow model. The ground water flow field obtained from the base case simulation was later used as a basis for the first tracer transport simulation. Therefore, a successful approximation of field conditions at well P26-M1 was essential to verify the applicability of the three-dimensional ambient flow simulation model. Furthermore, a streamline analysis was performed on the base case flow solution. Streamlines were plotted to analyze flow from a contaminant source for a grid layer located in the upper part of the flow model. The results of this analysis are presented in the tracer transport simulation results section.

The base case simulation was modified to generate modified flow simulations 1, 2, and 3. The modified simulations were used to study the influence of various model parameters on the ambient flow distribution. In modified case 1, the same aquifer and wellbore were used, but the porous medium was assumed to be homogeneous. Flow simulations for modified case 1 were then performed for three different hydraulic conductivity values. Another alternate simulation, modified case 2, was developed using the same homogeneous model domain, as used in modified case 1, but this time different vertical hydraulic gradients were applied to the model. In the last modified simulation, modified case 3, the thickness of the confined aquifer was reduced by 50% and 75% to show the effect of aquifer thickness on ambient flow distribution.

The first tracer transport simulation coupled the flow solution for the base case simulation (flow simulation for test well P26-M1) with the transport of a tracer, which was initially located in a high K zone of the lower portion of the aquifer. In addition, another flow solution was obtained for a model domain without a wellbore. This flow solution was used for a second tracer transport simulation with the same initial source strength and location as the previous transport run. The transport simulations were intended to demonstrate the movement of the tracer to other portions of the aquifer and to emphasize the dilution effect of ambient wellbore flow on the tracer concentration in and near the well. In the authors' opinion, the transport simulations demonstrated the ambient flow problem that would be associated with P26-M1 rather dramatically.

Numerical Model Domain Description

The flow equations were solved with a finite-difference method using MODFLOW-96 (Harbaugh and McDonald 1996). The grid was refined horizontally around the wellbore, i.e., the cells in the vicinity of the wellbore were finer. The horizontal size of the model domain was selected so that it was big enough to capture the effects of ambient flow in the proximity of the wellbore. Preliminary model runs indicated that the flow solution is unchanged if horizontal dimensions of the domain are larger than the currently selected dimensions. Figure 1 shows horizontal and vertical cross sections of the model domain used for the base case, modified case 1, modified case 2 flow simulations, and tracer transport simulations. The model domain for modified case 3 is not shown because grid dimensions were the same and only the number of layers was reduced.

The model for the base case simulation, modified cases 1 and 2 and for the transport simulations consisted of 59 columns, 59 rows, and 42 layers. The model domain had 42 layers because 42 borehole flowmeter measurements were taken for well P26-M1. The flow domain dimensions were 6.1 m$\times$6.1 m$\times$12.2 m. The grid was irregularly spaced in the horizontal (Δx_{min} = 0.02 m, Δx_{max} = 0.22 m; Δy_{min} = 0.02 m; Δy_{max} = 0.22 m). The horizontal refinement of the grid was necessary to capture the rapidly changing head gradients in the near proximity of the wellbore. Vertical layers were regularly spaced with Δz = 0.29 m. A smaller finite difference grid was set up for modified case 3. The aquifer thickness was reduced to 6.1 m and 3.05 m, resulting in modified model domains with 20 and 10 layers, respectively, instead of 42 layers. The model domains for modified case 3 had the same horizontal and vertical resolution as for the other cases.

The wellbore was represented as a group of gridblocks that stretched from the top to the bottom of the modeling domain. The wellbore cells were positioned at the center of the model domain. The simulation of the wellbore was accomplished by assigning a much higher hydraulic conductivity (K = 15000 m/day) to the cellblocks that represented the wellbore. Once the wellbore K was about three orders of magnitude greater than the aquifer K, flow to the well for fixed hydraulic conditions was controlled only by the aquifer K, so increasing the wellbore K further had no effect on the results.

The boundary conditions and the general setup of the model are shown in Figure 2. The upgradient and downgradient vertical boundaries of the domain were assigned a specified-head boundary condition, thus simulating a natural average horizontal hydraulic gradient. The lateral (along-gradient) boundaries were also defined as specified-head boundaries, but the specified head values were the same on both sides of the domain, and changed linearly from the upgradient to downgradient values. The top and bottom boundaries of the model domain were defined as specified-head boundaries, except for a small portion, which represented the uppermost and lowermost cross sections of the wellbore, where no vertical flow could occur. These gridblocks may be considered as no-flow boundaries and thus were represented in the model as no-flow gridblocks. The constant heads for each gridblock at the upgradient and downgradient vertical boundaries were assigned values gradually increasing in the z-direction, thus a natural vertical gradient ($\partial h/\partial z$) was established. The vertical gradient in the aquifer was expected to have an important effect on the magnitude of ambient flow.

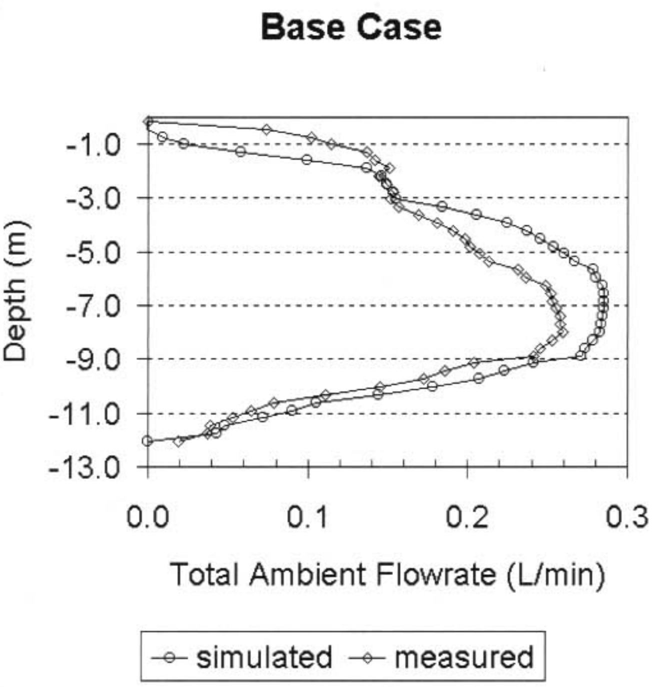
In order to establish the required uniform and natural horizontal gradient ($\partial h/\partial x$), slightly different head values (1.028 and 1.000 m) were used at the upgradient and downgradient boundaries. The constant head values assigned for the uppermost and lowermost gridblock layer and for the two lateral boundaries varied gradually in the x-direction. However, there was no gradient assigned in the lateral y-direction (normal to gradient); thus the x-axis was aligned along the direction of the horizontal gradient.

Selection of Realistic Parameters and Solution Procedure

Representative estimates for the natural hydraulic gradients were taken from a study by Watson (1998), who produced a potentiometric surface map of the Gordon Aquifer at the Savannah River Site near Aiken, South Carolina. The well cluster P26 is screened in the Gordon Aquifer, thus horizontal gradient values were estimated using the potentiometric surface map for this aquifer. The estimate for $\partial h/\partial x$ was 4.6×10^{-3}. In addition to that, vertical hydraulic head measurements and layer elevations were also known for well cluster P26. Information about the composition of the confining layers was made available from borehole experiments. From the material composition, an average hydraulic conductivity was estimated for the confining layers. Then the vertical hydraulic gradient, $\partial h/\partial z$, was calculated approximately by using head measurements and the estimated vertical hydraulic conductivity of the confining layers. $\partial h/\partial z$ was in the range from 1×10^{-3} to 1×10^{-4}. Thus a vertical gradient of 5.0×10^{-3} was selected for the base case flow simulation. No calibration or optimization of these directly estimated gradient values was performed.

Base Case Flow Simulation

Monitoring well P26-M1 that was modeled for the base case simulation was constructed to fully penetrate the Gordon Aquifer at the SRS site in Aiken, South Carolina. The well had an inner diameter (I.D.) of 15.25 cm, and the screen length was 12.2 m. Both the ambient flow distribution and the horizontal hydraulic conductivity (K_h) distribution for this well were measured and calculated, respectively, based on data from an electromagnetic borehole flowmeter (EBF) (Boman et al. 1997). The same horizontal hydraulic conductivity profile (Figure 3) obtained from the flowmeter tests by Boman et al. (1997) were used in the base case flow simulation. The EBF measurements were taken at 0.29 m intervals, therefore a total of 42 horizontal hydraulic conductivity values were given as the input K_h distribution for the model. The vertical hydraulic conductivity K_v was calculated from an assumed anisotropy ratio (K_h/K_v) of 10. For lack of better information, this ratio is commonly selected as a "rule of thumb." A smaller value would be expected to simply increase the ambient flow for a given K_h.



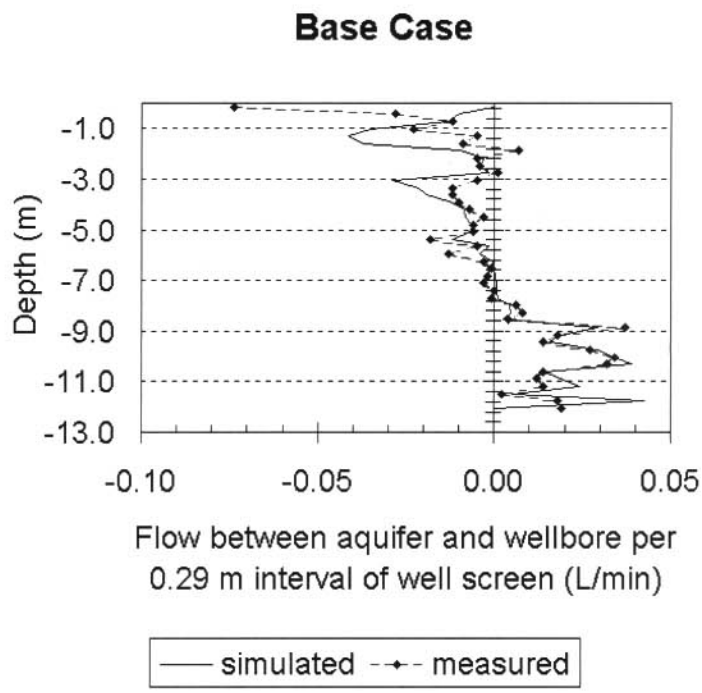


Figure 4. (a) Results for the base case simulation. Simulated total ambient flow compared to flowmeter results. Maximum ambient flow in simulated wellbore and measured ambient flow is about 0.4 m^3/day and 0.37 m^3/day, respectively. (b) Results for the base case simulation. Simulated differential ambient flow chart for well P26-M1 compared to measured ambient flow. The ground water enters the lower section of the well and exits from the upper section, resulting in upward wellbore flow.

Modified Flow Simulation Cases

For modified case 1, constant K_h values of 0.57, 5.7, and 57 m/day were used. K_v values were calculated from the anisotropy ratio of 10. The average hydraulic conductivity from the Boman et al. (1997) study was 5.7 m/day. Hydraulic gradients were the same as in the base case simulation. In modified case 2, the vertical gradient was altered and a constant K_h value of 5.7 m/day was applied. The vertical gradient is changed to 2.75×10^{-3} and 5×10^{-4}. The last modified flow simulation, modified case 3, was run for aquifer thicknesses of 6.1 m and 3.05 m, instead of the original thickness of 12.2 m., reducing the number of layers to 20 and 10, respectively. The average K_h of 5.7 m/sec and the same hydraulic gradients as in the base case simulation were retained. The base case model domain was used for all these modified cases, as shown in Figure 1, except for modified case 3.

The steady-state flow solutions for all flow simulations were obtained using the PCG2 (preconditioned conjugate gradient) solution method. Other solution methods that were tested included the strongly implicit procedure (SIP) and slice-successive over-relaxation (SSOR). Of these three options, the PCG2 had the lowest discrepancy in the overall mass balance and thus was selected as the solver for the flow simulations.

Ambient wellbore flow values were calculated by using the flow data from MODFLOW's CCF (cell-to-cell flow) output file. The net flow into and out of the wellbore was reported separately for each grid layer.

Tracer Transport Simulations

The first tracer transport simulation was built on the flow solution of the base case flow scenario. A constant concentration field of 100 mg/L was assumed to be located in the lower portion of the aquifer in a high K zone. The height, width, and depth of the constant concentration field were about 0.60, 0.14, and 11.6 m, respectively. Local longitudinal and transverse dispersivities were selected as 1 cm. This selection ensured that the transport problem was highly advection dominated, so it was necessary to select a solver that produced little numerical dispersion. In this case, a TVD (total-variation-diminishing) solution scheme based on the ULTIMATE algorithm (universal limiter for transient interpolation modeling of the advective transport equation) was selected. The third-order ULTIMATE scheme is mass conservative, without excessive numerical dispersion, and has essentially proven to be oscillation-free (Zheng et al. 1998). The second tracer transport simulation used a flow solution for the same model domain, but no wellbore was present. The hydraulic conductivity profile and natural hydraulic gradients from the base case flow simulation applied also to the transport simulations. The simulation time for the problem with the well was selected as 20 days, and the simulation time for the problem without a well was selected as 40 days.

Results of Simulations

Ambient Flow Simulations

Ambient flow simulation results were obtained by using the information from MODFLOW's CCF file. The CCF file contains flow rate values for each face of each gridblock. The flow entering or exiting the wellbore was calculated separately for each grid layer. The total ambient flow (vertical wellbore discharge) then could be calculated by summing the layer net flow values. The same calculation method was used for all flow simulations in this study.

The conceptual validity of the wellbore flow model, tied firmly to field measurements, was important, because it served as a basis for all the other flow simulation cases and the transport simulations. The first objective was to simulate the ambient wellbore flow of well P26-M1 at the Savannah River Site, which was previously measured by Boman et al. (1997). Figure 4 shows the results of this first simulation effort. Figure 4a shows the total ambient flow rate as a func-

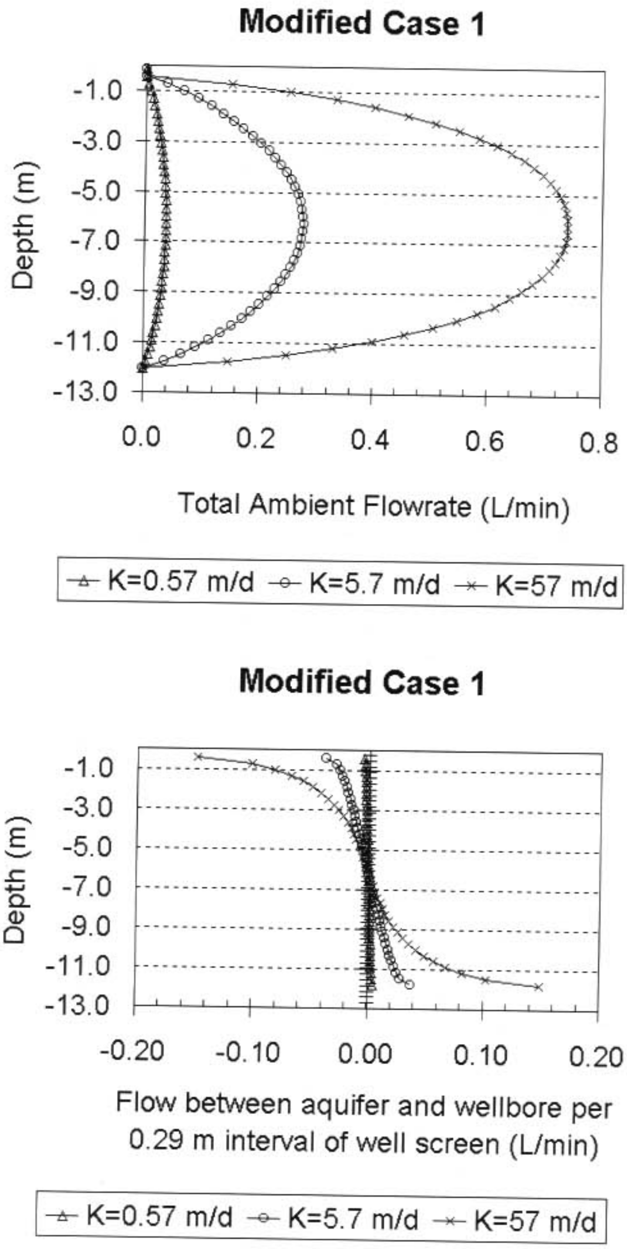


Figure 5. (a) Simulated total ambient flow in wellbore for modified case 1: simulation for three horizontal hydraulic conductivities. (b) Simulated differential ambient flow for modified case 1.

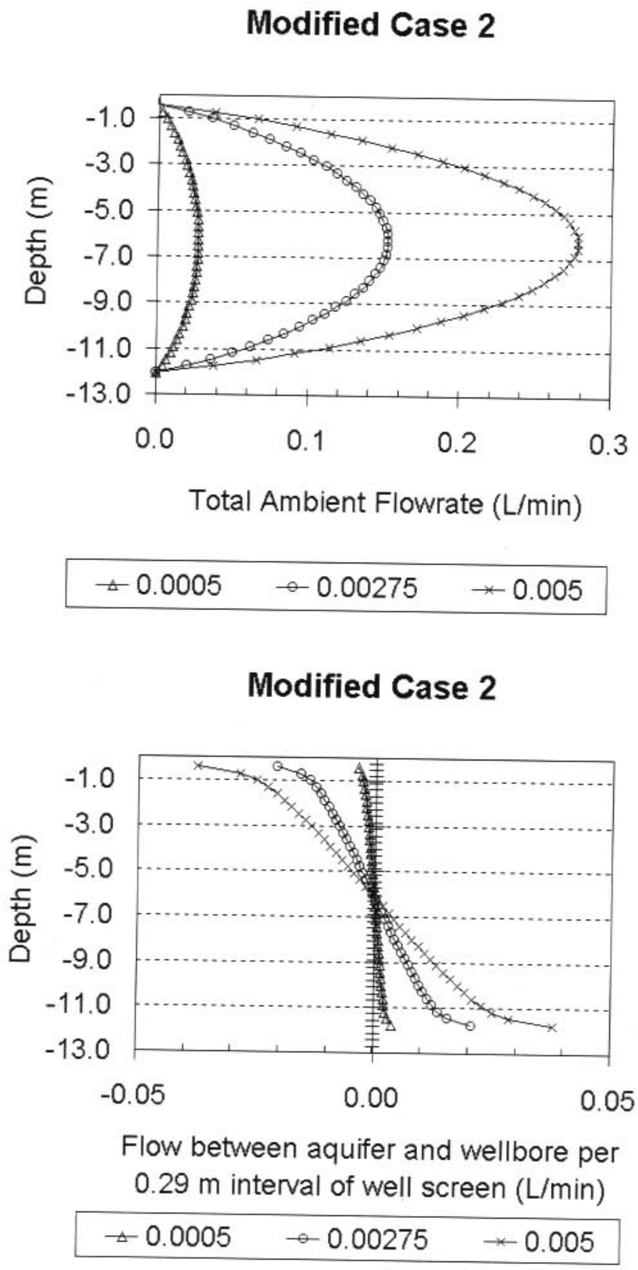


Figure 6. (a) Simulated total ambient flow in wellbore for modified case 2: simulation for three vertical gradients. (b) Simulated differential ambient flow for modified case 2.

tion of depth in the aquifer. A positive value of total ambient flow rate indicates upward flow. Overall the two curves agree well with maximum upward flows (0.259 L/min measured and 0.285 L/min simulated) occurring at about the same depth in the aquifer. Slightly modifying the vertical gradient or average K value could have been used to optimize agreement, but we chose not to do so. The calculated total ambient flow agreed well with the measurements, thus showing that the flow model well represented the field data. Flow between the aquifer and wellbore in each grid layer, sometimes called differential ambient flow (Molz et al. 1994), was obtained by differencing the curves in Figure 4a with the result shown in Figure 4b. Results after differencing provided a more stringent test of the model, but agreement was still good except in the upper third of the aquifer. Most likely, this disagreement was because of heterogeneity or errors in flowmeter measurements that were not accounted for in some way. In general, flow entered the well mainly through high K layers in the lower half of the aquifer and exited the well through similar layers in the upper part of the aquifer. This flow pattern resulted in an upward flow in the wellbore, as observed by Boman et al. (1997). Overall the results for the base case proved that the conceptual model is realistic and able to simulate ambient flow.

The results for modified case 1 (Figure 5) demonstrated that ambient flow is very sensitive to the horizontal hydraulic conductivity of the medium. The maximum ambient flow for a medium with an average homogeneous K_h of 5.7 m/day was about 0.28 L/min, which is almost equal to the result of the base case, from which the average was derived. The same simulation was run for a lower K_h of 0.57 m/day, one order of magnitude smaller than the

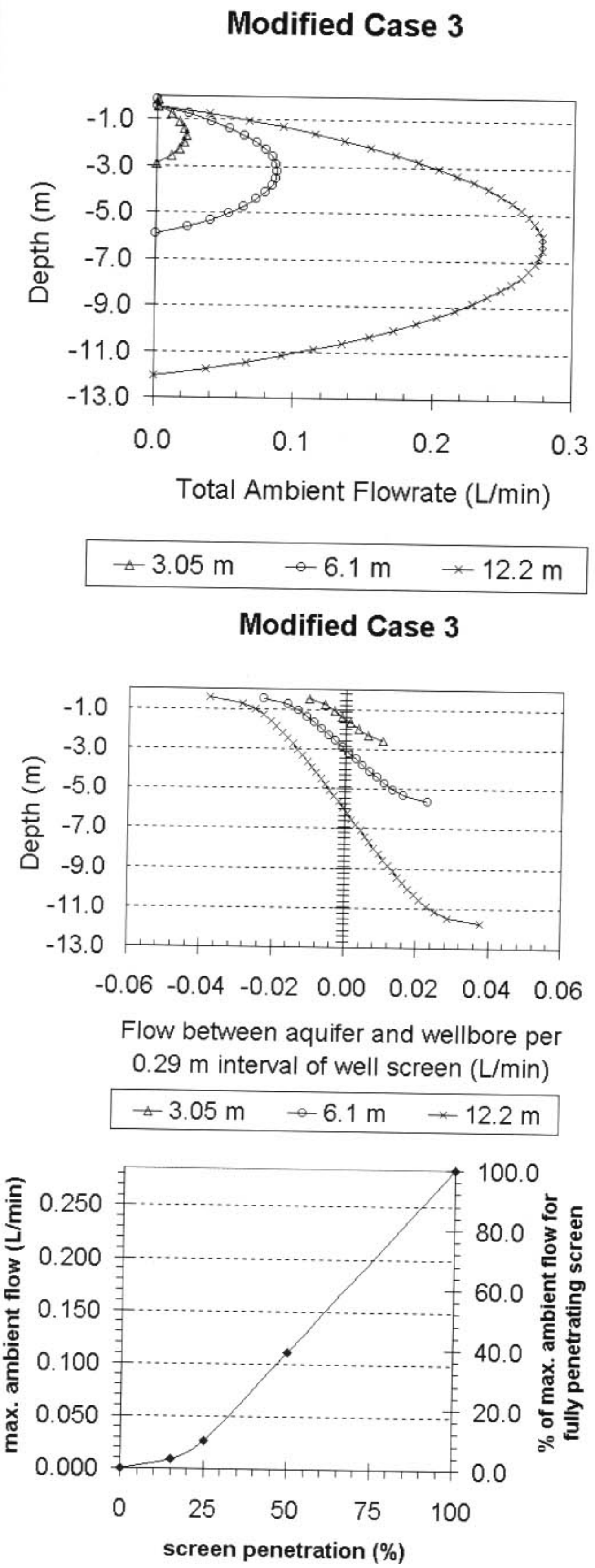


Figure 7. (a) Simulated total ambient flow in wellbore for modified case 3: simulation for three aquifer thicknesses. (b) Simulated differential ambient flow for modified case 3. (c) Change of maximum ambient flow with screen length. A 100% screen penetration implies a fully penetrating screen with a length of 12.2 m. The secondary y-axis shows the ratio of the maximum ambient flow rate to its fully penetrating value of 0.285 L/min, in percentage.

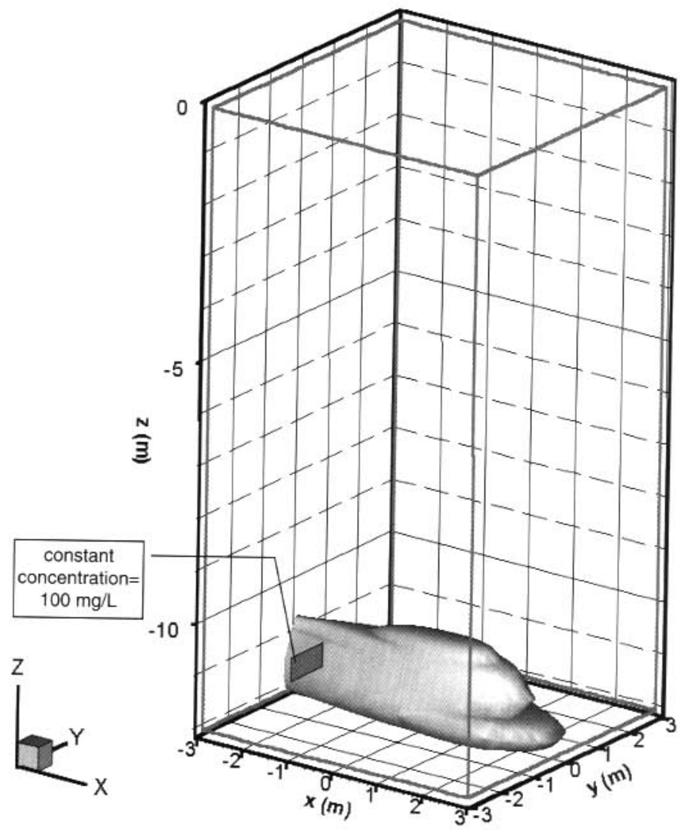


Figure 8. Iso-surface concentration plot of a simulated tracer in the aquifer without a wellbore. Plot shows position and size after 40 days simulation time. Arrows indicate main directions of ambient flow. Iso-surface shows 0.1 mg/L of tracer concentration.

average K_h, and a higher K_h of 57 m/day. The ambient flow rates for the lower K_h and the higher K_h were 0.038 L/min and 0.74 L/min, respectively. The anisotropy ratio K_h/K_v was selected as 10 for all three simulations. Ambient wellbore flow was more sensitive to K_h at lower K_h values. For more permeable media, K_h and K_v were still important factors in ambient flow, but were not as important as for a low K medium.

The average flow rate for the wellbore was about the same for both cases, the base case flow simulation with the measured K_h data set and the modified case with the average K_h value. The average flow rates for the base case flow simulation and the modified case with the average K_h were 0.179 and 0.183 L/min, respectively. The major difference between both cases was the distribution of inflow and outflow throughout the entire well screen. The average and maximum flow rates were comparable.

Results for modified case 2 are shown in Figure 6. In Figure 6b, a comparison of the differential ambient flow charts for three vertical gradients is shown. Figure 6a shows the comparison of total ambient flow for those vertical gradients. For this range of hydraulic gradients, the maximum ambient wellbore flow rate and the vertical hydraulic gradient were approximately proportional. The maximum ambient flow rate for a vertical gradient of 2.75×10^{-3} was 0.153 L/min, and for the smallest gradient of 5×10^{-4} it was 0.027 L/min, an order of magnitude smaller than flow with the original vertical gradient of 0.005. A vertical hydraulic gradient of 0.0005 would mean a head difference of 0.6 cm for an aquifer thickness of 12.2 m. This head difference is almost undetectable with conventional head measurement methods. The results clearly proved that even a very small vertical hydraulic gradient can cause a significant amount of ambient flow. The maximum ambient flow rate of 0.027

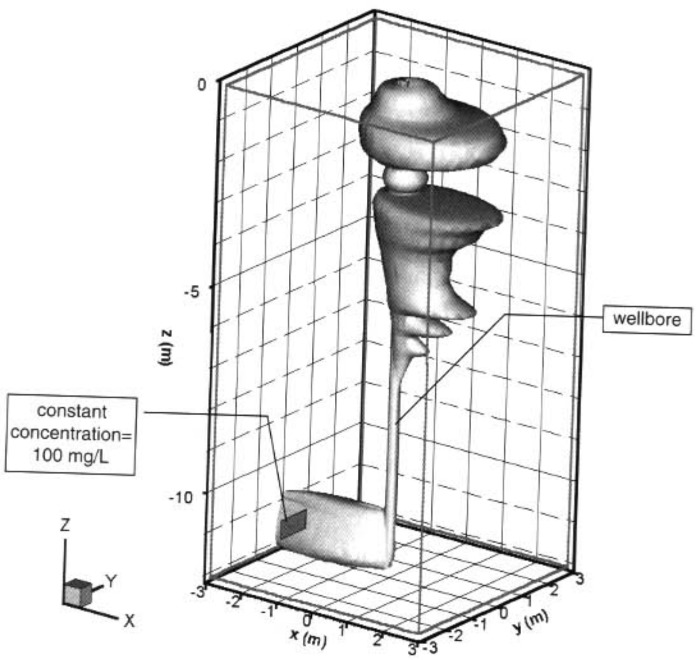


Figure 9. Iso-surface concentration plot of a simulated tracer in the aquifer with a wellbore. Plot shows position and size after 20 days simulation time. Arrows indicate main directions of ambient flow. Iso-surface shows 0.1 mg/L of tracer concentration.

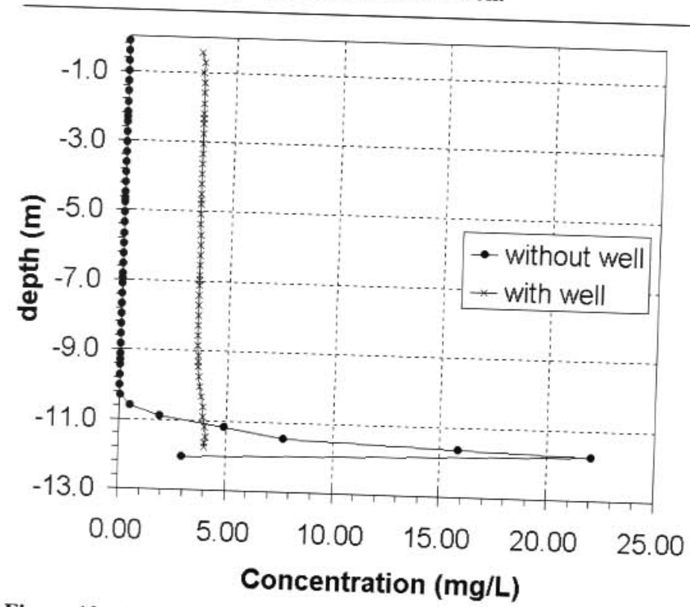


Figure 10. Comparison of concentration profiles at the center of the model domain (x = 0; y = 0). Profile for the simulation with well shows concentration change with depth within the well. Both profiles show concentrations for t = 20 days transport time.

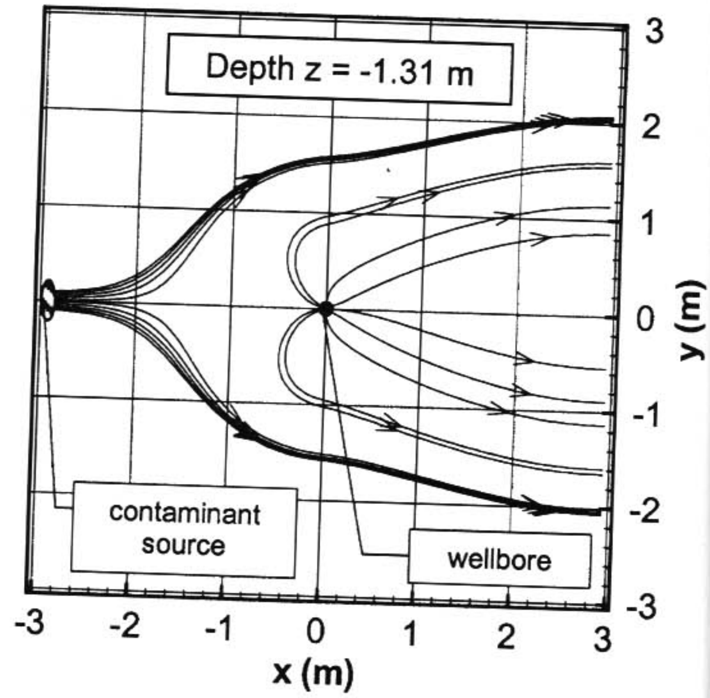


Figure 11. Streamline analysis for layer no. 5 from the top (z = − 1.31 m). Streamlines from a contaminant source located in the upper portion of the aquifer are bypassing the wellbore due to outward ambient flow.

L/min is above the threshold value of 0.01 L/min for the EBF (Molz and Young 1993). Heat pulse flowmeters have the capability of measuring even lower flows.

In modified case 3, the relationship between the aquifer thickness and the occurrence of ambient wellbore flow was studied. Results showed a clear correlation between aquifer thickness and ambient flow rate. Figure 7a shows that the maximum ambient flow rate for an aquifer thickness of 6.1 m was about 0.087 L/min, approximately 30% of the base value of 0.28 L/min. The aquifer thickness was reduced from 12.2 m to 6.1 m. An aquifer "size" reduction of 50% resulted in a flow reduction of approximately 70%. The maximum ambient flow rate for an aquifer thickness of 3.05 m was 0.021 L/min.

Screen length itself may be varied within an aquifer of constant thickness, which may be called the case of partial penetration, and such length changes would be expected to affect ambient flow. Figure 7c shows maximum ambient flow values, for the base case aquifer thickness of 12.2 m, as a function of screen length. (For all screen lengths, the screen is centered vertically about the midline of the aquifer.) Here an interesting result is evident. From full to about 25% penetration, the maximum ambient flow decreased linearly to about 10% of its fully penetrating value. The last 10% disappeared as the penetration decreased from 25% to 0%. Thus, the effect of shorter screen lengths can be quite significant in reducing the effect of ambient flows, although heterogeneity could still have an important and site-specific influence on such results. Although a thorough study of partial penetration effects is beyond the scope of the present paper, the preliminary results of Figure 7c suggest that screen lengths less than 25% of the aquifer thickness will produce much less ambient flow relative to the fully penetrating case. Of course, the effect of such flows may still not be negligible.

Tracer Transport Simulations

The effect of ambient flow on solute transport is made visible with the three-dimensional tracer transport simulation model. The flow result of the base case simulation (heterogeneous case) was used in the transport simulations, thus the same conceptual model domain was applied. The transport model was run for two scenarios: an aquifer without a wellbore, and an aquifer with a wellbore in place. Shown in Figures 8 and 9 are the dramatic results of the transport simulations. Both illustrations show iso-surface plots of the concentration distributions after a selected time. The iso-surfaces represent a concentration of 0.1 mg/L. For the second scenario, a simulation time of 20 days was long enough to see the movement of the tracer to upper portions of the aquifer solely because of ambient flow. The wellbore functioned here as a preferred flow channel, which caused the tracer to disperse over the entire well length. Without the wellbore, the tracer remained in the lower portion of the aquifer and none of it entered the upper regions. The transport simulation results were also in qualitative agreement with the field observations made by Hutchins and Acree (2000). They observed a dilution effect during sampling, thought to arise from preferential flow of recharge water from the water table to deeper portions of the contaminated zone that they monitored. The numerical results presented here also indicated a dilution effect because of spreading of the tracer plume over the entire well screen (see Figure 9).

The ambient flow pathway for water movement can result in several effects, including a dilution of ground water sampled by a long-screened well. Such dilution can lead to misinterpretation and overestimation of ground water remediation efforts. This effect is demonstrated in Figure 10. Vertical concentration profiles were extracted from both transport simulation results and they were compared with each other. The profiles show the concentration change with depth at the location of the wellbore. It should be noted that one profile represents the concentration change for the model domain without the well. In the absence of a well, the maximum detectable concentration in the vertical was about 23 mg/L. On the other hand, if there was a well in place, a significantly lower concentration would be seen. The peak concentration in this case was about 4 mg/L and was detected in the lower part of the monitoring well. The location of the peak concentration is at the intersection of the tracer plume with the well. The concentration in the remainder of the well was decreased until it reached a value of about 3.50 mg/L. The rapid change in concentration in the lower part of the well was due to the inflow of fresh ground water, resulting in significant dilution. The concentration change in the upper part of the well was smaller because the net flow of ground water in this part of the well was from the well to the aquifer.

Another effect of wellbore flow was the significant displacement of the tracer plume. In the absence of ambient wellbore flow, the tracer would not be detected in the upper portion of the aquifer. The displacement of a certain amount of water volume occurred in this case for a vertical gradient of 5×10^{-3} and would also occur for a gradient that is an order of magnitude smaller (5×10^{-4}), as shown in modified case 2. The dilution of the contaminant plume into the top half of the aquifer is also clearly visible from Figure 9.

It was evident also from the streamline analysis that a nearby contaminant source in the upper portion of an aquifer (Figure 11) containing a wellbore with upward ambient flow could be missed entirely. This effect is illustrated in Figure 11 for the predicted flow field in the fifth layer (at $z = -1.31$ m depth) from the top of the base case simulated herein. The outward flow (note streamline directions) from the top half of the well would divert flow of the contaminant and cause it to bypass the monitoring well. Whenever one sampled the well, one would be sampling water only from the lower portions of the aquifer where the ambient flow originates. Well purging would not help because of the large cumulative volumes of ambient flow surrounding the upper portion of the wellbore, such as 12 m^3/month for the base case.

Conclusions and Recommendations

Measurable ambient wellbore flow will likely occur in the majority of aquifers. This paper presented the results of simulated flow and transport in the vicinity of a monitoring well, and related such flows to common hydraulic conditions and aquifer parameters in the near-well vicinity. A solid tie to reality was maintained by using field data from a monitoring well in a heterogeneous aquifer near Aiken, South Carolina, to test predictions of a flow model successfully. Results supported the conclusion that sampling processes from long-screened monitoring wells will not be reliable, and often not even interpretable. A contaminant plume in a particular stratum, commonly a zone of higher K, that is approaching a monitoring well can be short-circuited through the wellbore, spread out, and mixed with water in-flowing from other strata. The mixture will then be expelled to some other higher K strata. The results from sampling such a well will commonly be diluted samples and a concentration distribution that has an ambiguous relationship to the true concentration distribution in the plume being monitored. In some cases a contaminant source in the near-well vicinity would not be detected at all if it happened to be located in a zone with ambient out flow from the monitoring well. Even though ambient flow is rather small, it occurs continuously, so that large cumulative volumes result over time.

The occurrence of significant ambient wellbore flow should be expected on many scales. This conclusion follows from the results of modified case 3, where a 50% and 75% smaller aquifer thickness was tested. The screen length itself had a significant effect in reducing the magnitude of ambient flow. The results suggest that screen lengths less than 25% of the aquifer thickness will produce much less ambient flow relative to a fully penetrating well. Moreover, the effect of the vertical hydraulic gradient on the magnitude and the distribution of ambient flow was very significant. It should be kept in mind that vertical gradients are dynamic in nature and magnitudes can change over time. The direction and magnitude of gradients depend on the type of aquifer and also on recharge conditions. In this study it was assumed that the vertical hydraulic gradient was constant throughout the entire depth of the aquifer. However, the direction of the gradient could be reversed in one region of the aquifer, which will cause a mixed ambient flow pattern, instead of an upward or downward flow only.

The maximum upward flow for the base case studied herein was about 0.28 L/min, which would cause an induced exchange of approximately 12 m^3/month of ground water per month from the bottom half of the aquifer to the upper half. As long as there is a vertical hydraulic gradient, this exchange is expected to happen continuously and thus purging of this well prior to sampling would have a minimal effect.

Our results were consistent with previous field experiments that documented a bias in ground water sampling because of ambient flow (Church and Granato 1996; Hutchins and Acree 2000). Finally, we must agree with the early warning of Reilly et al. (1989) to abandon or phase out the use of long-screened monitoring wells for ground water sampling. Short-screen cluster wells screened at different elevations of the aquifer, or multilevel samplers, should be used instead. In fact, it is likely that the last two decades of ambiguous data from long-screened monitoring wells has contributed to a widespread "homogenized" view of plume heterogeneity. Fortunately, this view is changing, and multilevel sampling wells are finding increased use for water quality monitoring.

Acknowledgment

This research has been supported in part by a grant from the U.S. Environmental Protection Agency's Science to Achieve Results (STAR) program. However, it has not been subjected to any EPA

review and therefore does not necessarily reflect the views of the agency, and no official endorsement should be inferred.

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