ALAMEDA COUNTY
HEALTH CARE SERVICES
AGENCY
DAVID J. KEARS, Agency Director

ENVIRONMENTAL HEALTH SERVICES
ENVIRONMENTAL PROTECTION
1131 Harbor Bay Parkway, Suite 250
Alameda, CA 94502-6577
(510) 567-6700
FAX (510) 337-9335

July 8, 2005

Mr. Kevin Graves
State Water Resources Control Board
Underground Storage Tank Cleanup Unit
PO Box 2231
Sacramento, CA 95812

Dear Mr. Graves:

Subject: Petition Response to Fuel Leak Case RO0000516, Hooshi's Auto Service, 1499 MacArthur Boulevard. Oakland, CA 94602

Alameda County Environmental Health (ACEH) has prepared this letter in response to the "Petition for Closure" dated May 6, 2005, prepared by Cambria Environmental Technology, Inc. (Cambria). Cambria the environmental consultant for Ms. Naomi Gatzke, requests case closure for the above referenced facility. Based on their review of the site background and conditions, Cambria believes that this site meets the Regional Water Quality Control Board - San Francisco Bay Region's (RWQCBs) definition of a "low-risk fuel site" as defined in their memorandum *"Interim Guidance on Required Cleanup at Low-Risk Fuel Sites"*, dated January 5, 1996. Cambria has attached their "*Closure Request,"* dated July 21, 2004 and their letter titled "*Clarifications Regarding Closure Request,"* dated October 6, 2004, as supporting documentation for their closure petition.

ACEH has reviewed the data for the subject site and Cambria's closure request and finds Cambria's arguments lacking technical basis and therefore reject their claims. Our responses to each of Cambria's arguments regarding their site's compliance with the various criteria for a low-risk groundwater case as defined in the RWQCB's memorandum are discussed below.

1. Criterion 1: The Leak Has Stopped and Ongoing Sources, Including SPH, Have Been Removed

ACEH agrees that leaks of petroleum hydrocarbons have stopped since the underground storage tanks were removed in 1990 and no new USTs have been re-installed at this site. However, a compelling demonstration that SPH has been completely removed from the subsurface has not been made. SPH was indeed present at this site as evidenced by the presence of up to 1-foot of free product in wells MW-2 and MW-5.

Cambria installed and operated a SVE system in MW-1, MW-2, and MW-5. When the SVE system failed to remove significant mass Cambria performed "air sparging using the vacuum from the SVE blower" in MW-2 and MW-5. The system(s) operated for 8-months and removed only 16.5 pounds of hydrocarbons. Cambria subsequently analyzed groundwater samples from the same monitoring wells they performed remediation activities in and presented these results as demonstrating the effectiveness of their remediation.

However, our evaluation of the remediation efforts in the context of the site geologic and hydrogeologic conditions indicates that the remediation efforts were largely ineffectual. Cambria performed remediation from September 2000 to April 2001 when the water table was 6.9 to 8.8-feet bgs. This level is above the top of the well screens in wells MW-1 and MW-2 that were ostensibly being used for SVE. That means that the screens in MW-1 and MW-2 were completely submerged during the SVE operations, making it impossible to recover any petroleum vapors from those two wells. The well screens in well MW-5 extend higher than the other two wells (approximately 5 feet higher than the piezometric surface at the time when SVE operations occurred) which could have made it a better candidate for SVE. However, the portion of the screens in MW-5 that are above the water surface are entirely within an interval where a clay aquitard exists. Consequently, little petroleum vapors would likely have been extracted from that interval either. We note that the sand pack in MW-5 extends up into a shallow sand that is less than 4 feet below the ground surface. Thus, it seems likely that some – or even all – of the vapors extracted by Cambria during their SVE operations at this site may have been from a very shallow, largely uncontaminated sand right next to MW-5 that is several feet above the intervals containing the SPH or high levels of contamination. Air recharge to the MW-5 would be primarily from the atmosphere (i.e., short-circuiting) given the shallow depth of the sand pack in MW-5. It is therefore no surprise that after only one month of operation the SVE system ceased to remove any hydrocarbon vapors. It is not clear at all that any SPH was removed from the subsurface at this site as a result of Cambria's SVE operations.

We are also unclear about the details of the air sparging operation described in Cambria's Closure Request. There is a description of air sparging in MW-2 and MW-5 (i.e., wells containing SPH) using "the vacuum from the SVE blower." We assume that Cambria meant that the positive-pressure side of the vacuum blower was used to inject air into these wells. This concerns us. Injecting air into a well containing free product could drive the product out of the well and further into the formation. Thus, the "sparging" operation could make the problem worse. Further, we do not know at what depth in the wells the air was injected into. Wells MW-2 and MW-5 both have well screens that extend at least 7 feet below a clay aquitard that exists beneath the site (see discussion of the geologic site characterization below). If air was injected into the sand formation beneath the aquitard, it may have become trapped beneath the overlying aquitard (since SVE operations were incapable of extracting any vapors other than those at very shallow depths next to Well MW-5 as described above). The injected air could have displaced dissolved contaminants and potentially mobilized vapors that could have emerged or "daylighted" tens or hundreds of feet away from MW-2 and MW-5 (depending on the lateral extent and vertical permeability of the aquitard). This situation could have posed a potential explosion hazard or unacceptable exposure to vapor receptors.

Finally, Cambria never performed any verification sampling to see if SPH still exists in the formation (i.e., away from the wells). They note that SPH is no longer detected in MW-2 and MW-5 (perhaps because Cambria pushed the SPH out of the wells during the sparging operation?). However, Cambria cannot conclude whether or not it occurs in the formation between the wells because verification soil sampling has not been performed. Monitoring data obtained from wells used for remediation merely measures the treatment installed at the remediation point and in no way demonstrates the effectiveness of remediation within the aquifer. Consequently, compliance with Criterion 1 has not been met.

2. *Criterion 2: The Site Has Been Adequately Characterized*

With regard to the geology beneath the site, there has been a lot of information collected, but the information has not been thoroughly interpreted or integrated into a coherent site conceptual model (SCM). Consequently, geologic aspects of the SCM cannot be considered to be well characterized. For example, there are no geologic cross-sections included in Cambria's technical reports that depict the subsurface conditions. Nor are the details of the tank excavations(s), including depths, backfill compositions, etc., clearly described and depicted graphically. Definition of the site geology is crucial for understanding pathways for contaminant migration, the likelihood of focused recharge, remediation system design, etc. For example, there is an aquitard beneath the site at depths ranging from 4 to 12-feet based on borehole data. This isn't described in the reports presented by Cambria but is likely a very important feature that may impede deeper migration of contaminants or, perhaps, the upward migration of vapors. Further, Cambria and other site consultants were either unaware of this unit or ignored it when they installed many of the site monitoring wells. This resulted in the installation of several monitoring wells that straddle the aquitard and connect the permeable units above and below the aquitard (e.g., MW-4, MW-5 and MW-6). Consequently, the groundwater monitoring network may be ineffective and some wells may even contribute to cross-contamination of the water-bearing zones beneath the site. In addition, a thorough understanding of the site geology is needed to design an effective remediation program. It appears that the site geologic conditions were not adequately considered in the design of the SVE and air sparging system (see discussion in Comment 1).

The site hydrogeologic conditions are also poorly defined. In particular, the groundwater flow field is not well characterized. The groundwater contour map included in the Closure Petition depicts equipotential lines drawn using water levels in site monitoring wells measured on April 2, 2004. The horizontal groundwater flow direction inferred from this figure is toward the southeast. Cambria states in their Petition that MW-4 monitors a dissolved plume downgradient from the source zone based on the flow directions inferred from the groundwater contour map. However, we have reviewed several other groundwater contour maps prepared using water level data on different dates (Exhibit 3) and horizontal groundwater flow directions implied by the equipotential lines in those maps suggest flow to the northeast. Some or most of the variability in the inferred horizontal groundwater flow direction could be attributed to hydraulic mounding in the center of the site. If there is mounding that is causing diverging flow, this mounding can also cause downward vertical flow. This could be a mechanism whereby contaminants in or near the tank backfill could have been conveyed to depths greater than the total depth explored or monitored at the site. Cambria has not addressed this question, nor can they with the existing groundwater monitoring network since there are no well pairs or clusters installed at the site that could yield important information regarding the vertical distribution of dissolved contaminants or hydraulic head. There have been no hypotheses formulated to explain the mounding nor the causes of the (related?) erratic/variable apparent groundwater flow directions. That is a fundamental deficiency in the site characterization because an effective groundwater monitoring program cannot be developed until the site groundwater flow system is defined in three dimensions.

Regarding the source of groundwater contamination, the presence of SPH in the subsurface is evidenced by SPH being detected in two monitoring wells (MW-2 and MW-5). However, the lateral and vertical extent of SPH has not been defined. Thus, the remediation system installed at this site could not have been effective at completely removing SPH from the subsurface beneath the site. Unfortunately, the effectiveness of the remediation system at removing SPH from the subsurface cannot be evaluated because verification sampling was never performed, as discussed above.

The vertical extent of soil contamination at the source is undefined. Soil samples collected beneath the three former fuel USTs in 1990 detected up to 450 milligrams per kilogram (mg/kg) TPHg and 8.7 mg/kg benzene. There was no documentation that any of the residual contaminated soil below the sampled depth was removed. Soil sample results from borings, MW-2 and G-9, collected at comparable depths to those collected beneath the USTs and located at the edge of the excavation indicate that higher contaminant concentrations may be expected at deeper depths beneath the USTs. During the installation of MW-2, 1,460 mg/kg TPHg at 10-feet below ground surface (bgs) was detected in 1993. In 1996, soil sample results from G-9 detected 860 mg/kg TPHg and 3.1 mg/kg benzene at 12.5-feet bgs and was the deepest soil sample collected from this boring leaving the vertical extent of contamination undefined.

The lateral and vertical extent of the dissolved contaminant plume(s) is also undefined. There are only two monitoring wells installed along the property boundary. Given the extremely variable groundwater flow directions that may exist at this site, that number of monitoring points is inadequate. In spite of water level contour maps suggesting flow radiating from the center of the site, contaminant delineation efforts have been primarily to the southwest. Since groundwater elevation data suggests multiple and changing groundwater flow directions, additional plume delineation in other flow directions should have been performed.

The discussion of plume definition so far has focused on horizontal components of flow. No vertical definition of the contaminant plume has been performed. Vertical definition of the contaminant plume is necessary for the site to be considered "adequately characterized." Vertical assessment of dissolved contamination at this site should utilize depth-discrete groundwater sampling tools.

In conclusion, based on the aforementioned technical issues, ACEH does not consider this site to be adequately characterized. Therefore, it does not comply with Criterion 2

3. Criterion 3: The Dissolved Hydrocarbon Plume Is Not Migrating

As discussed above, site characterization and monitoring performed to date has been inadequate to ensure that dissolved contaminants are not migrating off site without being detected. If the monitoring program cannot be relied on to ensure detection of dissolved contamination emanating from the facility, it clearly cannot be relied on to determine whether or not the leading edge of the dissolved hydrocarbon plume has stabilized and is not longer migrating, which is the requirement of Criterion 3.

Cambria states that concentrations of TPHg and benzene will decrease to concentrations below their respective ESLs by 2010 based on simple trendlines drawn on graphs showing concentrations of TPHg and benzene over time in well MW-2. This analysis is invalid for the following reasons:

- Separate phase hydrocarbons (SPH) were consistently observed in well MW-2 prior to 2002. Several concentrations of TPHg that are included on the time concentration graph for well MW-2 exceed the expected normal range of solubility for gasoline. The intermittent incorporation of small volumes of SPH in groundwater samples has a significant effect on TPHg and benzene concentrations. It is not valid to project a linear trend over time using initial concentrations that exceed the solubility of gasoline. The decreases in concentrations over time are artifacts that reflect the incorporation of less SPH in the groundwater samples and clearly cannot be projected to continue at a linear rate over time.

- If only the data that represent dissolved phase concentrations are plotted (April 2002 to April 2004), a linear trendline drawn through the data shows an increasing trend.

- However, we disagree that linear extrapolations like the ones performed by Cambria are reliable predictors of future concentrations of dissolved contaminants. The primary reason for this is that Cambria has not described the mechanism whereby the concentrations will be attenuating over time. There are several reasons why the rates of chemical and/or biochemical reactions my decrease over time, with the result that plots of declining concentrations over time reach asymptotic levels.

Based on the above factors, Cambria's projection of concentration decreases over time based on TPHg and benzene data from well MW-2 is seriously flawed and invalid.

Cambria has not demonstrated that hydrocarbon concentrations in groundwater are decreasing on-site; nor that decreasing hydrocarbon concentrations in groundwater indicate that natural attenuation is remediating the site hydrocarbons; nor that the plume in groundwater is shrinking. Cambria suggests that data from MW-4 (potentially occasionally downgradient) and ND well MW-6 (that is not within the plume) as evidence of natural attenuation and thus plume shrinking. A review of data from the other wells at the site suggests otherwise and indicates that hydrocarbon concentrations are increasing. Also, decreasing hydrocarbon concentrations in groundwater is not sufficient to demonstrate that natural attenuation is remediating the site hydrocarbons. Natural attenuation would need to be demonstrated by several lines of evidence, such as measurement of by-products, consumption of electron acceptors, etc.

Cambria uses the hydraulic conductivity of 1 to 2.6 X 10-5 centimeters per second (cm/s) [Century West Engineering Corporation (CWEC), 1996] and the thickness (12-feet) of the clayey sands to indicate that groundwater flow and chemical migration is restricted and confined to the site and does not appear to be migrating offsite. However, the contaminant concentrations in groundwater detected away from the source area, i.e. MW–3, suggests that groundwater flow and chemical migration can migrate through these soil materials and possibly move offsite.

Some of the data presented in the Closure Request Report suggests that significant biases may occur in the data sets Cambria used to support their recommendations regarding case closure. For example, in the graph titled "TPHg and Benzene Concentrations Trend Well MW-5" (Exhibit 1), there is a strong cyclic fluctuation in TPHg and benzene concentration that appears to be seasonal in nature beginning in 2000. The site experienced a rapid rise in groundwater in 2000 specifically during the rainy season. As water level raised contaminant concentration decreased. We have drawn lines on this graph to indicate a rainy season of November to May. During these months, and allowing for a delay for recharge of the aquifer to occur, contaminant concentrations in MW-5 decrease during this time period as water level rises. This trend suggests that a significant negative bias exists in these data due, perhaps, to ambient vertical flow of recharge water in the well due to recharge-induced downward vertical hydraulic gradients (see Exhibit 2 for further discussion of this topic). Consequently, these data are not suitable to support the claims of successful remediation of the aquifer nor natural attenuation or biodegradation of contamination made by Cambria.

In conclusion, we do not believe that the data presented by Cambria to date show in any way that dissolved hydrocarbon plume(s) at this site have stabilized and are no longer migrating. Thus, Cambria has not shown that this site is in compliance with Criterion 3 (the dissolved hydrocarbon plume is no longer migrating).

4) Criterion 4: No Water Wells, Deeper Drinking Water Aquifers, Surface Water, or Other Sensitive Receptors are Likely to be Impacted

Cambria did not perform a complete preferential pathway study for this site. Groundwater has been as shallow as 5-feet bgs and the NAPL and dissolved phase hydrocarbon plumes could have encountered migration pathways and potential conduits (including sewers, storm drains, pipelines, trench backfill, etc) that can spread contamination away from the site yet, this pathway was not evaluated. Cambria's preferential pathway study is incomplete and not in accordance with 23 CCR Section 2654(b) as they failed to evaluate the utility pathway.

Additionally, Cambria performed a 250' door-to-door survey of beneficial use wells. The rationale for the limited radius selected by Cambria and for omitting the review of well log records at the Department of Water Resources (DWR) offices, Alameda County Public Works, and other sources was not provided. Cambria's well survey is not in accordance with 23 CCR Section 2654(b) and the Regional Board's guidance for identification, location, and evaluation of potential deep well conduits.

For the reasons stated above Cambria has failed to demonstrate compliance with Criterion 4.

5) Criterion 5: The Site Presents No Significant Risk to Human Health or the Environment

To assess the potential health risks to occupants of the site and adjacent property, Cambria compared site hydrocarbon concentrations with the ESLs. Recent data indicates that the highest TPHg and benzene concentration detected was in well MW-2 at 37,000 micrograms per liter (ug/l) and1,200 ug/l, respectively, substantially exceeding the ESLs. Cambria uses TPHg and Benzene Concentration vs. time graphs for a single monitoring well, MW-2 in which they have drawn decreasing trendlines (through data showing an increasing trend) to extrapolate a date (2010) by which these compounds would be below their respective ESLs. Cambria uses this evaluation as justification for case closure today. For reasons stated in Comment 3, Cambria's projection of concentration decreases over time based on TPHg and benzene data from well MW-2 is seriously flawed and invalid.

Cambria states that because the plume is shrinking and is not expected to extend from the site, there is no significant risk to surface water, wetlands or other ecological receptors. However, Cambria has not demonstrated that the plume is shrinking, nor that it has remained onsite (as discussed in Comment 3). Further, we have concerns that the remediation performed at the site by Cambria may have caused further migration of SPH, dissolved plume, and vapors from this site (as discussed in Comment 1).

An additional health risk element that was not considered by Cambria was migration of contamination via pathways other than the dissolved phase. Therefore, in addition to offsite migration of dissolved contaminants via preferential pathways; a second migration pathway, vapor migration (affecting human health & safety), must be investigated and evaluated and/or in the vicinity of the site.

ACEH notes that up to 8.7 ppm benzene was left in place in shallow soil at the site in addition to 1,200 ppb benzene presently in groundwater at depths of 5-feet bgs. Additionally, the fate of the SPH at this site is unknown (as discussed in Comment 1). Therefore, resultant migration of vapors, in particular the more toxic constituents of gasoline, emanating from residual contamination and LNAPL at the site could pose an inhalation risk at and/or in the vicinity of the site.

Thus Cambria has failed to demonstrate compliance with Criterion 5.

CONCLUSIONS

Cambria has failed to demonstrate that this site meets any criteria for case closure including the RWQCB criteria for a low-risk fuel site. for the reasons discussed in this letter. Moreover, the concentrations of TPHg and benzene in samples collected during the most recent quarter (21,000 ug/L TPHg and 400 ug/L benzene in Q2 2005) are substantially higher than the ESL values, even for groundwater that is not a source of drinking water (500 ug/L TPHg and 46 ug/L benzene). Consequently, we recommend that Cambria develop and implement additional site assessment efforts and remediation measures that address the deficiencies in the SCM outlined in this letter, and which will reduce petroleum hydrocarbon concentrations beneath the site to target ESLs in a reasonable time frame.

If you have any questions, please call Mr. Don Hwang at (510) 567-6746.

Sincerely,

Don Hwang

Don Hwang
Hazardous Materials Specialist

Donna L. Drogos

Donna L. Drogos, P.E.
LOP Program Manager

Enclosures -

Exhibit 1 – "TPHg and Benzene Concentration Trend Well MW-5," from Cambria reported dated July 21, 2004 with ACEH additions

Exhibit 2 – "Implications of Observed and Simulated Ambient Flow in Monitoring Well," Elci, Molz, and Waldrop

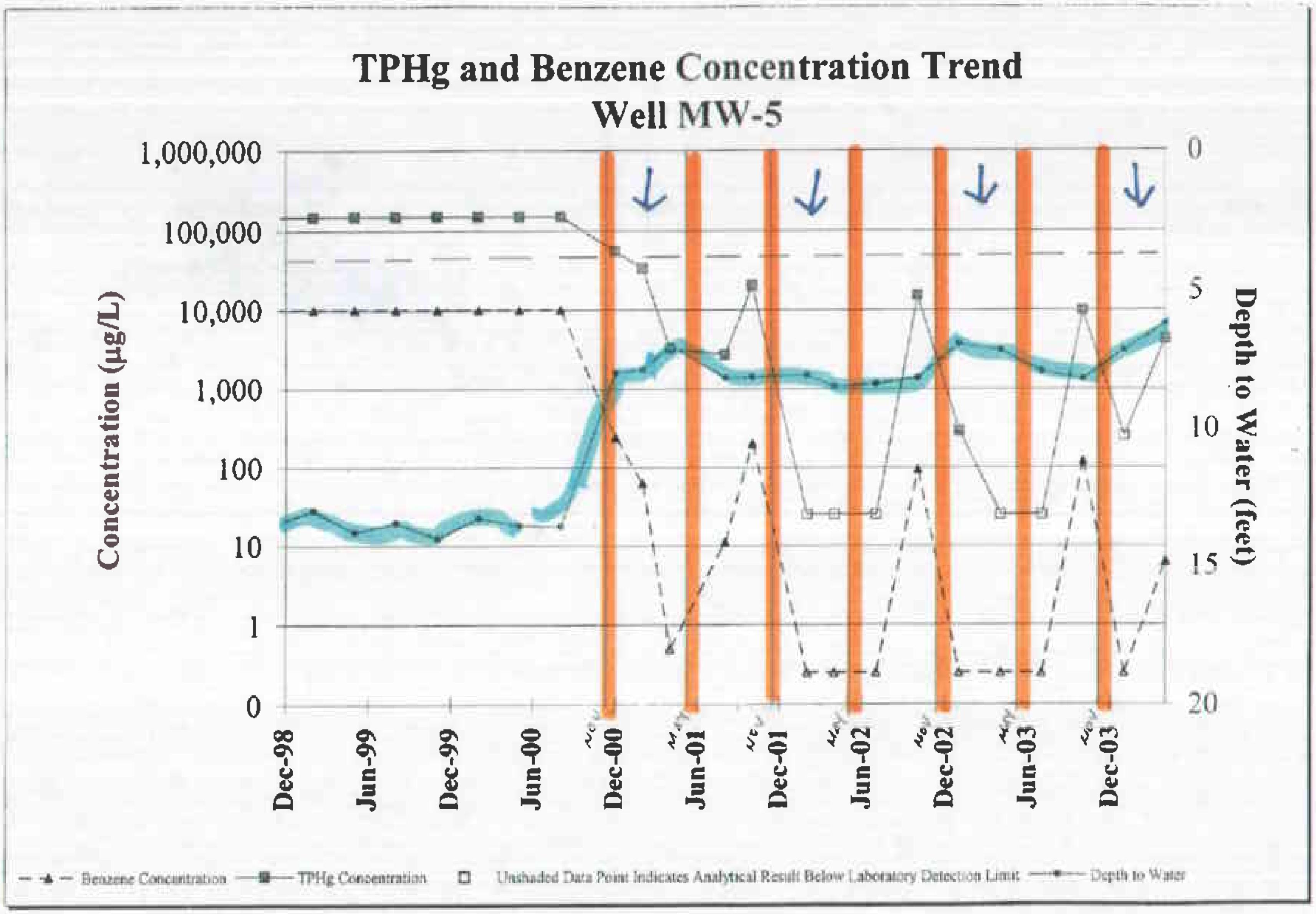
Exhibit 3 – Cambria groundwater elevation contours, 05/08/1998, 02//17/99, and 0/17/00

cc: Ms. Liz Haven (w/enc), State Water Resources Control Board, UST Cleanup Unit, PO Box 2231, Sacramento, CA 95812

Ms. Naomi Gatzke (w/enc), 1545 Scenic View Drive, San Leandro, CA 94577

✓ Mr. Matthew Meyers (w/enc), Cambria Environmental Technology, Inc., 1144-65th Street, Suite B, Oakland, CA 94608

A. Levi (w/enc), D. Drogos (w/enc), D. Hwang (w/enc)



CAMBRIA –(REPORT DATED JULY 21, 2004)

EXHIBIT 1

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

[1]Environmental Engineering and Science, Clemson University, 342 Computer Ct., Anderson, SC 29625; (864) 656-1009; fax (864) 656-0672; aelci@clemson.edu

[2]Environmental Engineering and Science, Clemson University, 342 Computer Ct., Anderson, SC 29625; (864) 656-1003; fax (864) 656-0672; fredi@clemson.edu

[3]Quantum Engineering Corp., 112 Tigitsi Ln., Loudon, TN 37774; (865) 458-0506; fax (865) 458-0504; wrwaldrop@aol.com

Received September 2000, accepted April 2001.

EXHIBIT 2

Table 1
Summary of Documented Ambient Wellbore Flow[1]

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)	[2]Louisiana BAP1; 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
	[2]Mobile, AL (deeper well); confined	mixed	20	1.75
Hutchins and Acree (2000)	[2]Eglin AFB, FL; 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

[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.

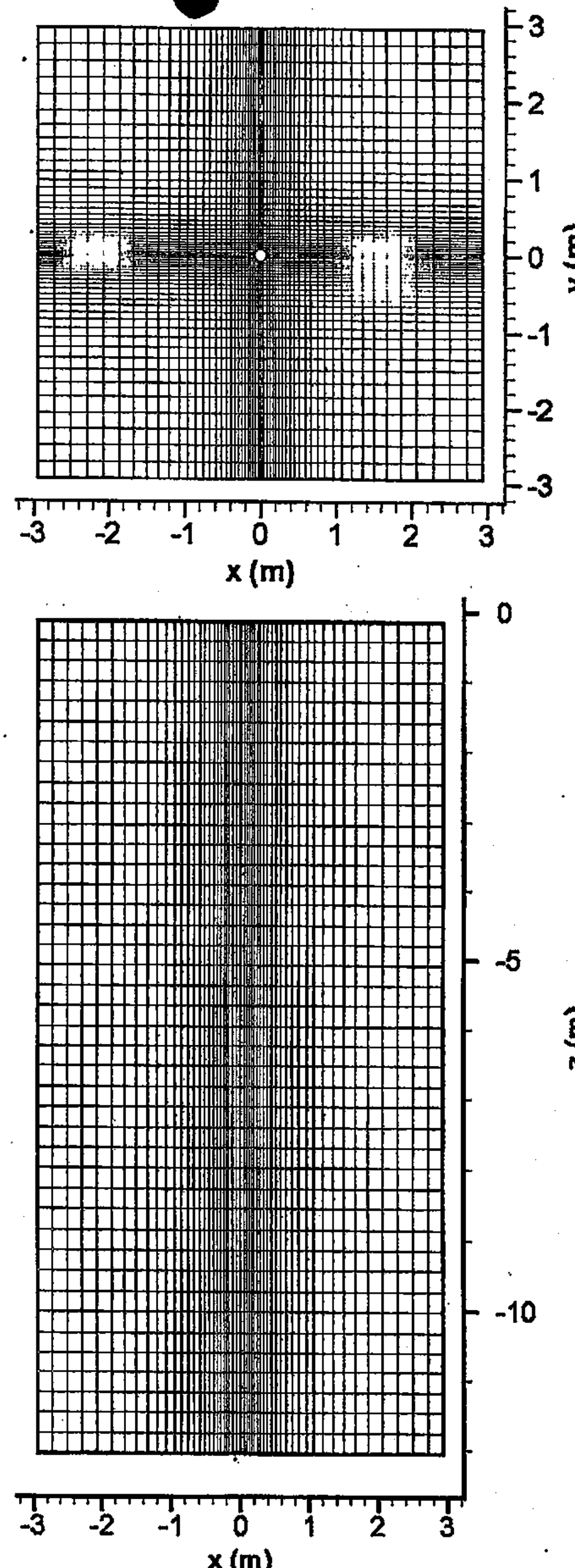


Figure 1. Finite difference grid used to simulate ambient wellbore flow. This grid was used for base case simulation, modified case 1, modified case 2, and the transport simulation. The grids for modified case 3 had 10 and 20 layers; the horizontal dimensions were the same. (a) plan view; (b) side view.

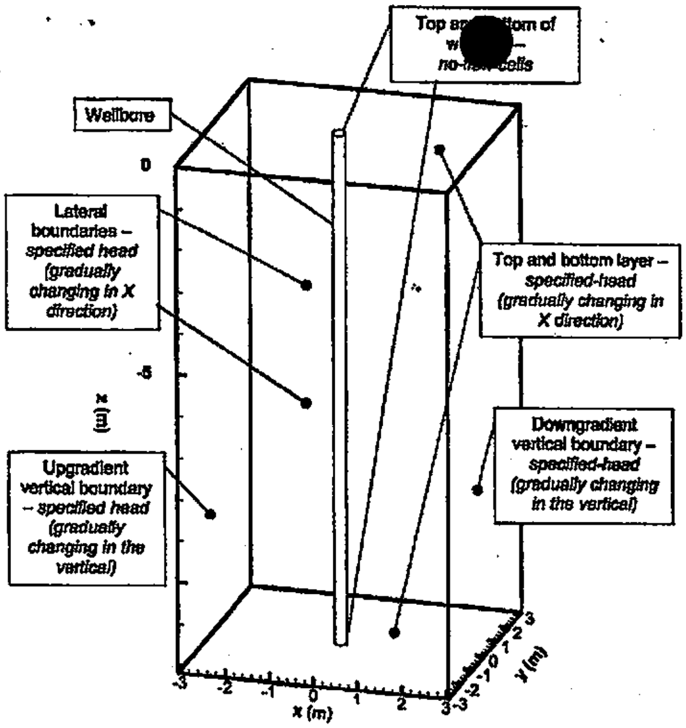


Figure 2. Diagram showing boundary conditions of the model.

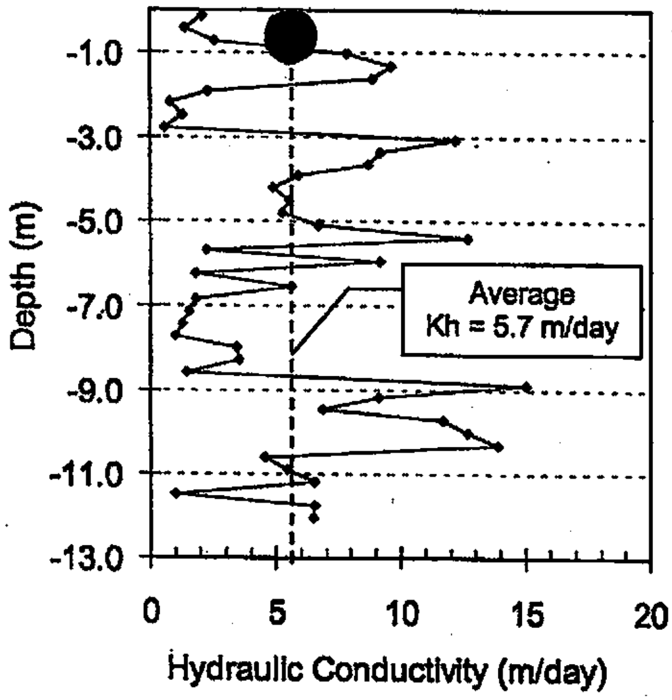


Figure 3. Hydraulic conductivity distribution by flowmeter analysis for well P26-M1 (Boman et al. 1997).

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 ($\Delta x_{min} = 0.02$ m, $\Delta x_{max} = 0.22$ m; $\Delta y_{min} = 0.02$ m; $\Delta y_{max} = 0.22$ 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 $\Delta z = 0.29$ 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.

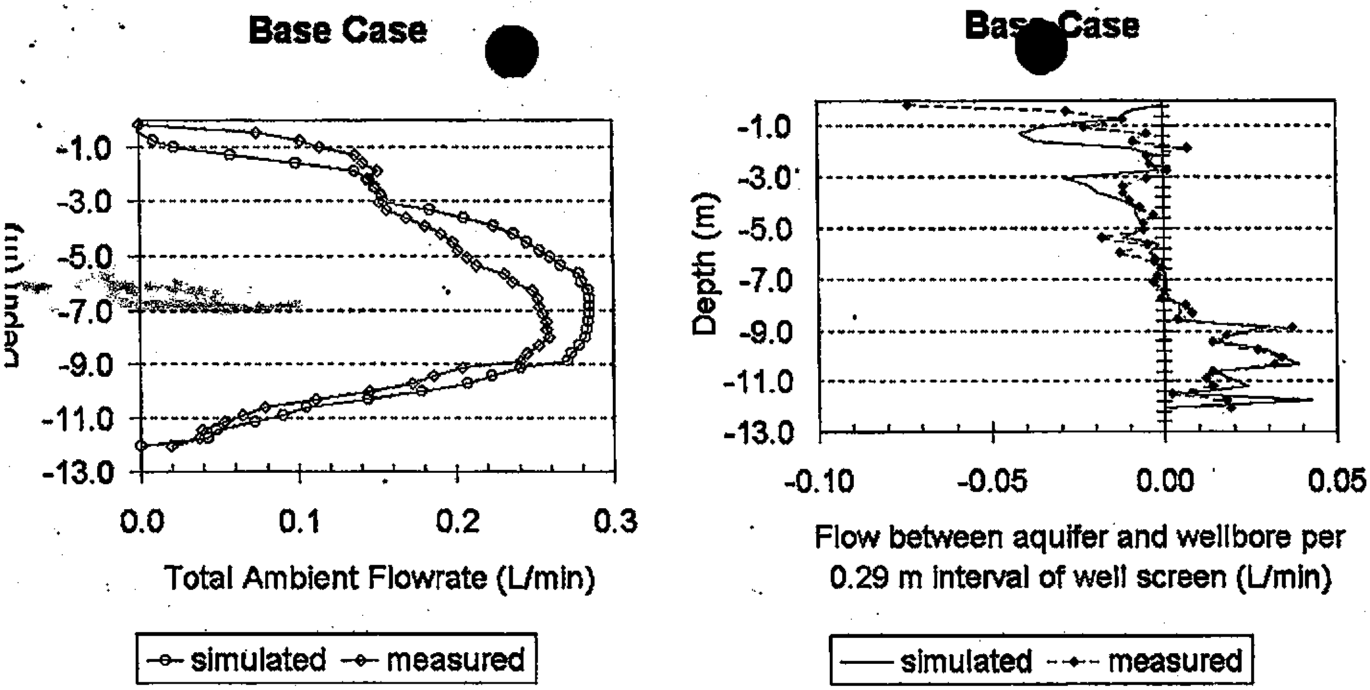
In order to establish the required uniform and natural horizontal gradient ($\partial h/\partial x$), slightly different head values (1.028 and 1.000 m) were used at the upgradient and downgradient boundaries. The constant head values assigned for the uppermost and lowermost gridblock layer and for the 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.



gure 4. (a) Results for the base case simulation. Simulated total ambient flow compared to flowmeter results. Maximum ambient flow in sim- ated wellbore and measured ambient flow is about 0.4 m³/day and 0.37 m³/day, respectively. (b) Results for the base case simulation. Simulated fferential ambient flow chart for well P26-M1 compared to measured ambient flow. The ground water enters the lower section of the well id exits from the upper section, resulting in upward wellbore flow.

lodified Flow Simulation Cases

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

The steady-state flow solutions for all flow simulations were btained using the PCG2 (preconditioned conjugate gradient) solu- on method. Other solution methods that were tested included the trongly implicit procedure (SIP) and slice-successive over-relax- tion (SSOR). Of these three options, the PCG2 had the lowest dis- repancy in the overall mass balance and thus was selected as the olver for the flow simulations.

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

Tracer Transport Simulations

The first tracer transport simulation was built on the flow olution of the base case flow scenario. A constant concentration ield of 100 mg/L was assumed to be located in the lower portion f the aquifer in a high K zone. The height, width, and depth of the onstant concentration field were about 0.60, 0.14, and 11.6 m, espectively. 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 well-bore 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-

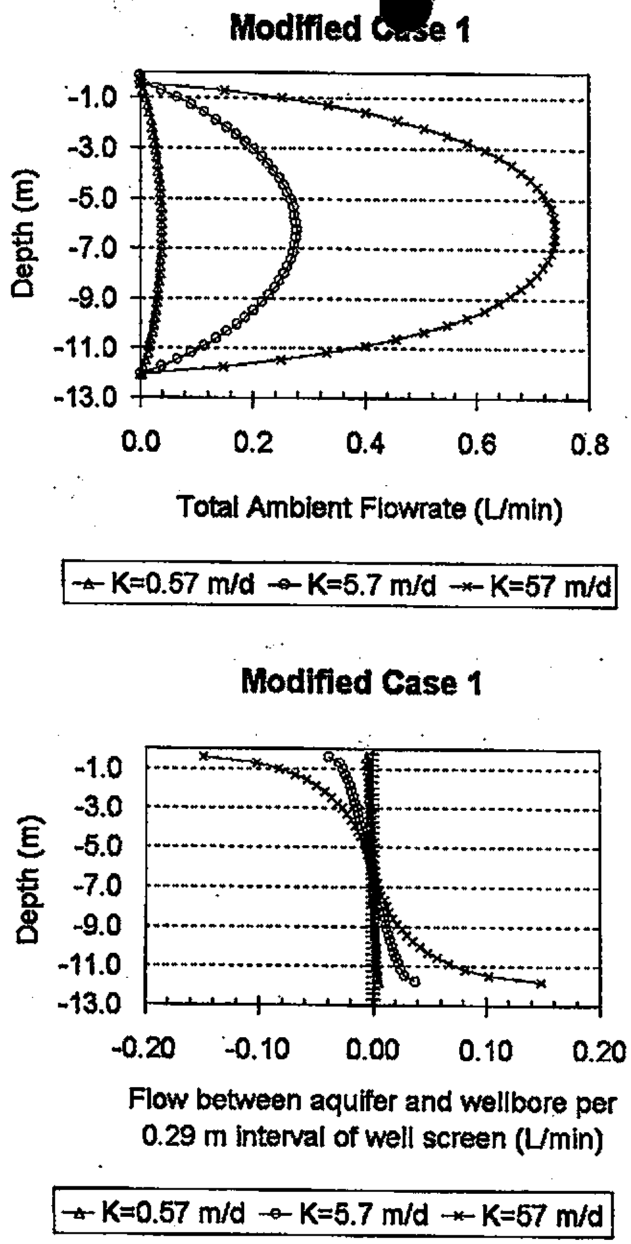


Figure 5. (a) Simulated total ambient flow in wellbore for modified case 1: simulation for three horizontal hydraulic conductivities. (b) Simulated differential ambient flow for modified case 1.

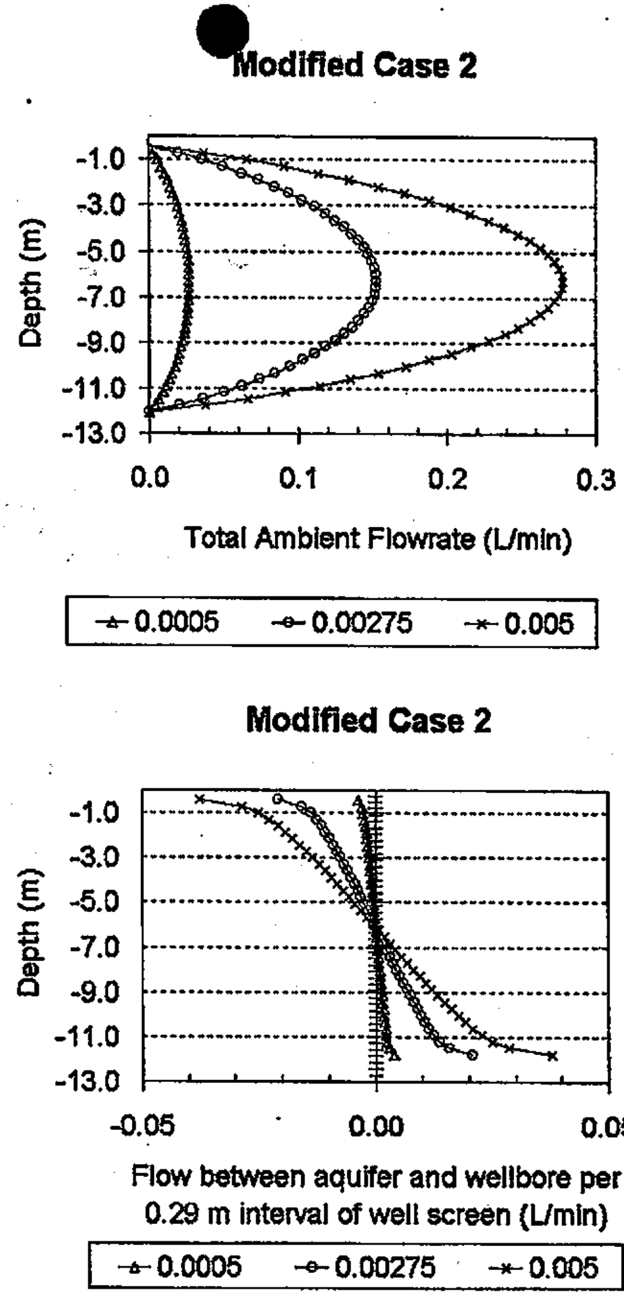


Figure 6. (a) Simulated total ambient flow in wellbore for modified case 2: simulation for three vertical gradients. (b) Simulated differential ambient flow for modified case 2.

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

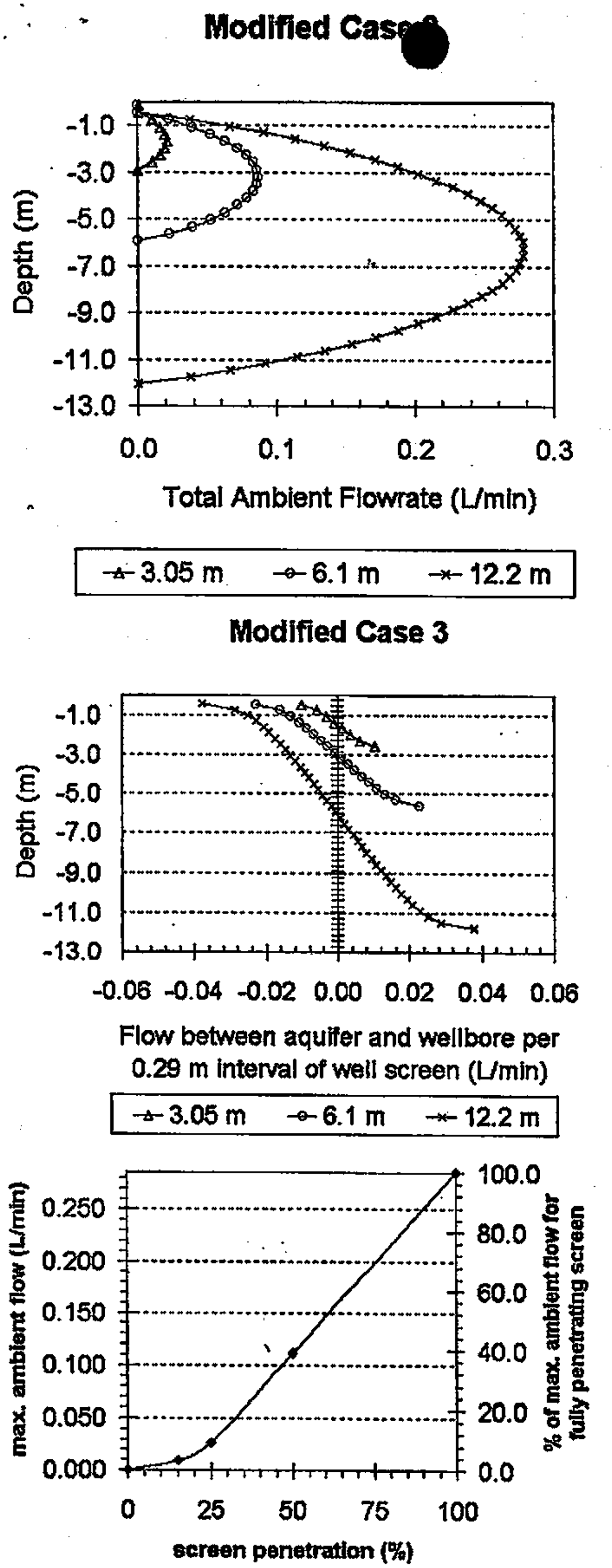


Figure 7. (a) Simulated total ambient flow in wellbore for modified case 3: simulation for three aquifer thicknesses. (b) Simulated differential ambient flow for modified case 3. (c) Change of maximum ambient flow with screen length. A 100% screen penetration implies a fully penetrating screen with a 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.

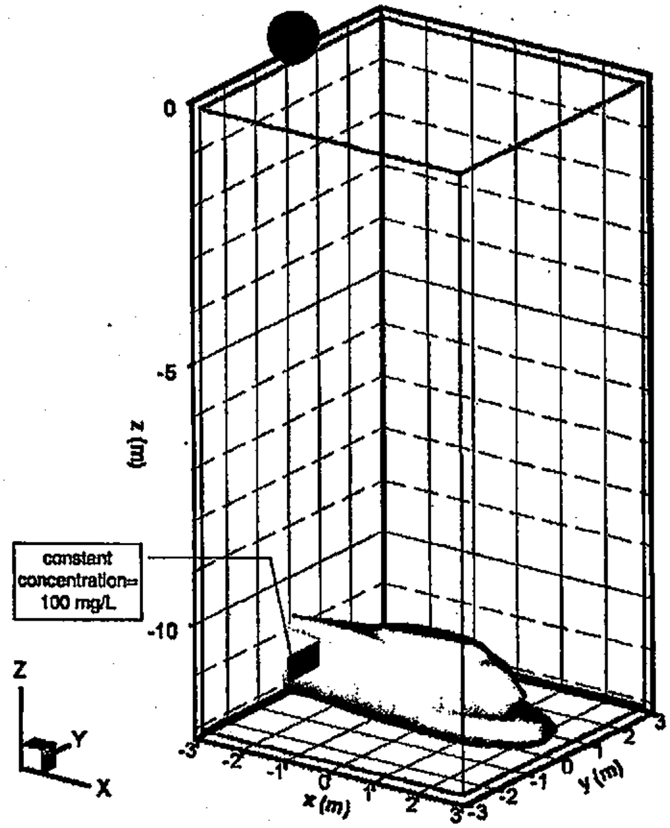


Figure 8. Iso-surface concentration plot of a simulated tracer in the aquifer without a wellbore. Plot shows position and size after 40 days simulation time. Arrows indicate main directions of ambient flow. Iso-surface shows 0.1 mg/L of tracer concentration.

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

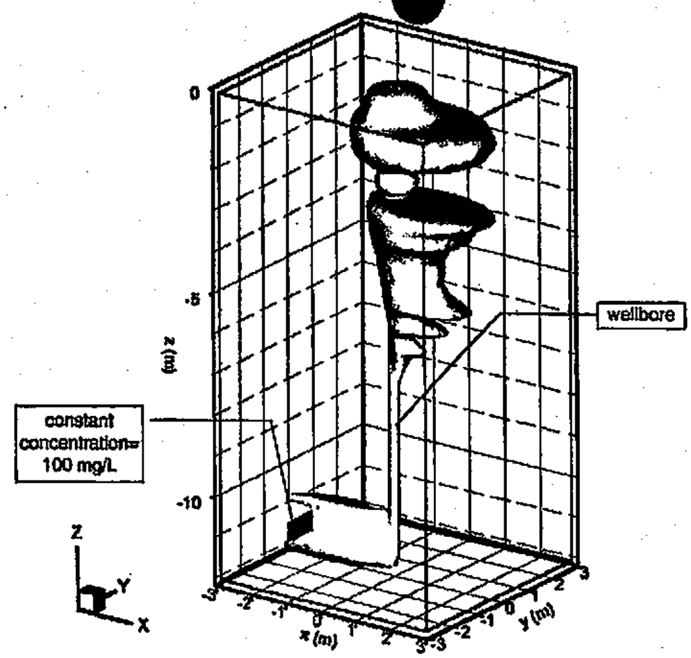


Figure 9. Iso-surface concentration plot of a simulated tracer in the aquifer with a wellbore. Plot shows position and size after 20 days simulation time. Arrows indicate main directions of ambient flow. Iso-surface shows 0.1 mg/L of tracer concentration.

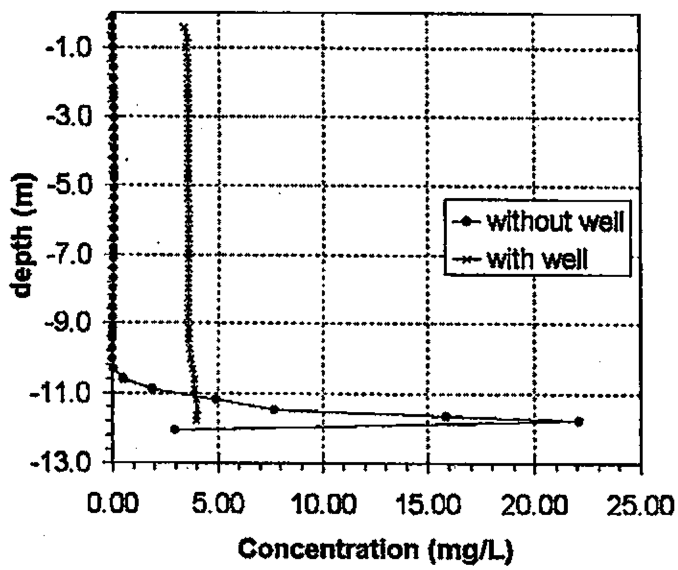


Figure 10. Comparison of concentration profiles at the center of the model domain (x = 0; y = 0). Profile for the simulation with well shows concentration change with depth within the well. Both profiles show concentrations for t = 20 days transport time.

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.

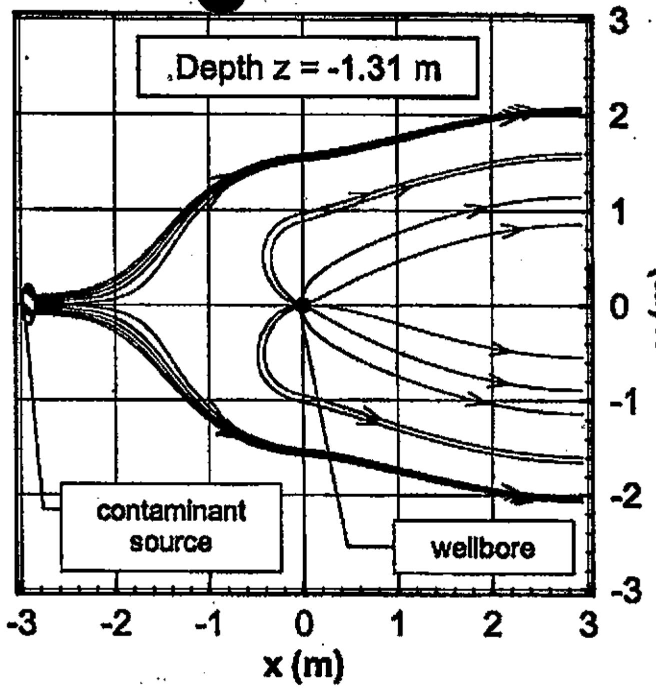


Figure 11. Streamline analysis for layer no. 5 from the top (z = − 1.31 m). Streamlines from a contaminant source located in the upper portion of the aquifer are bypassing the wellbore due to outward ambient flow.

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

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

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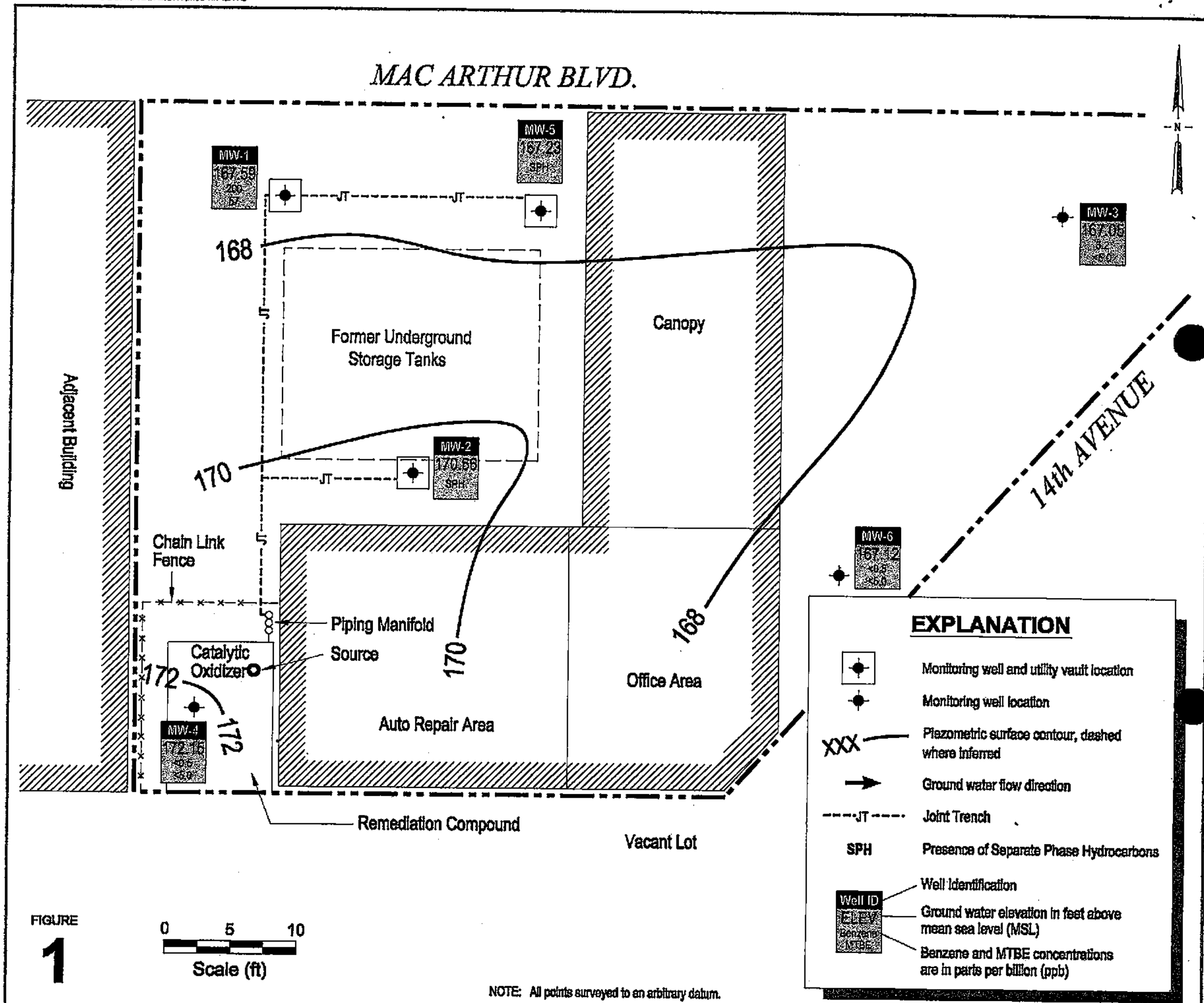

Hooshi's Auto Service
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Oakland, California

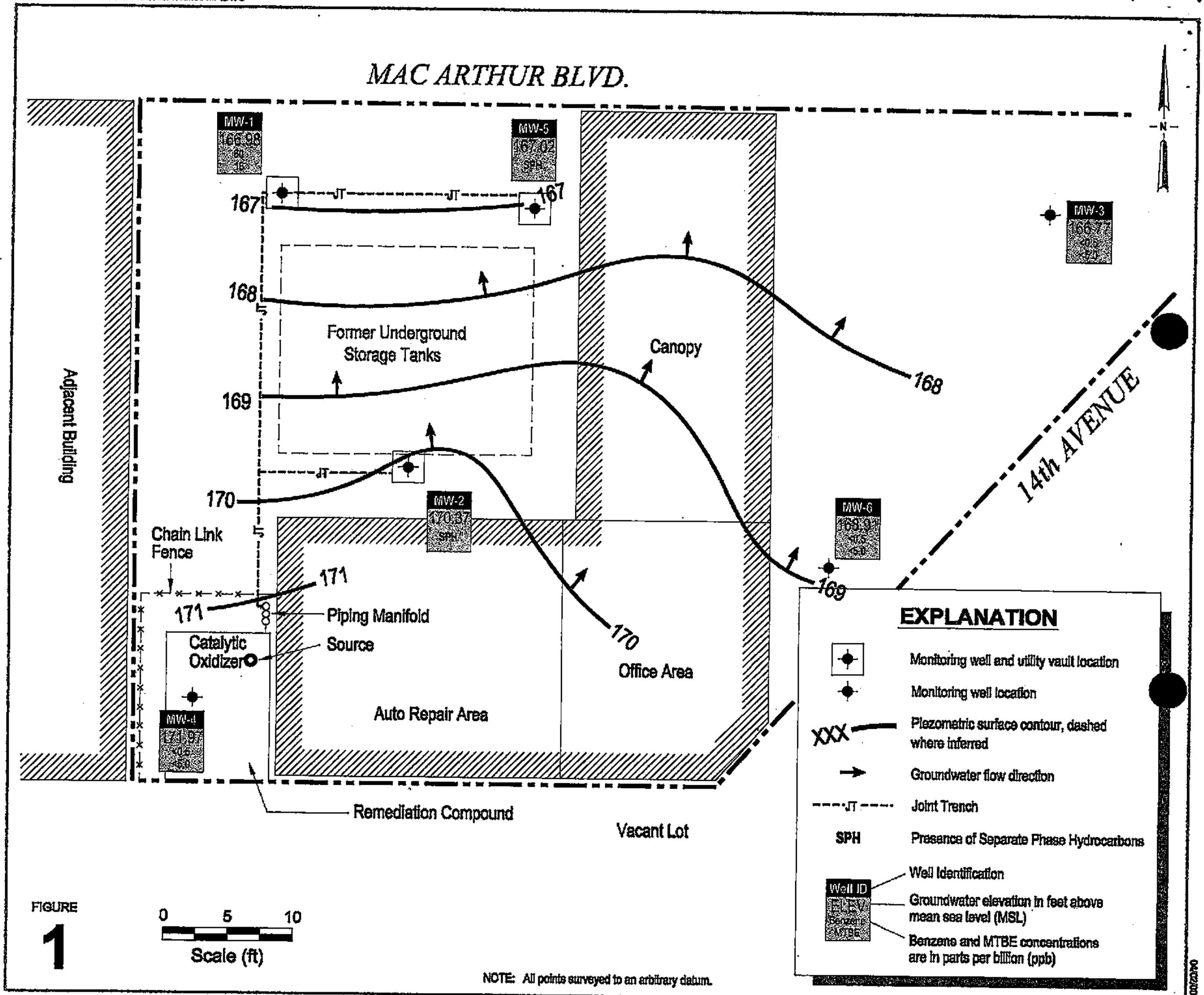
CAMBRIA

Ground Water Elevation Contour Map
February 17, 1999

FIGURE
1

EXHIBIT 3

H:\SB-2004\OAK-HOOSHI\FIGURES\1CM00-MP.DWG



FIGURE

1

Hooshi's Auto Service

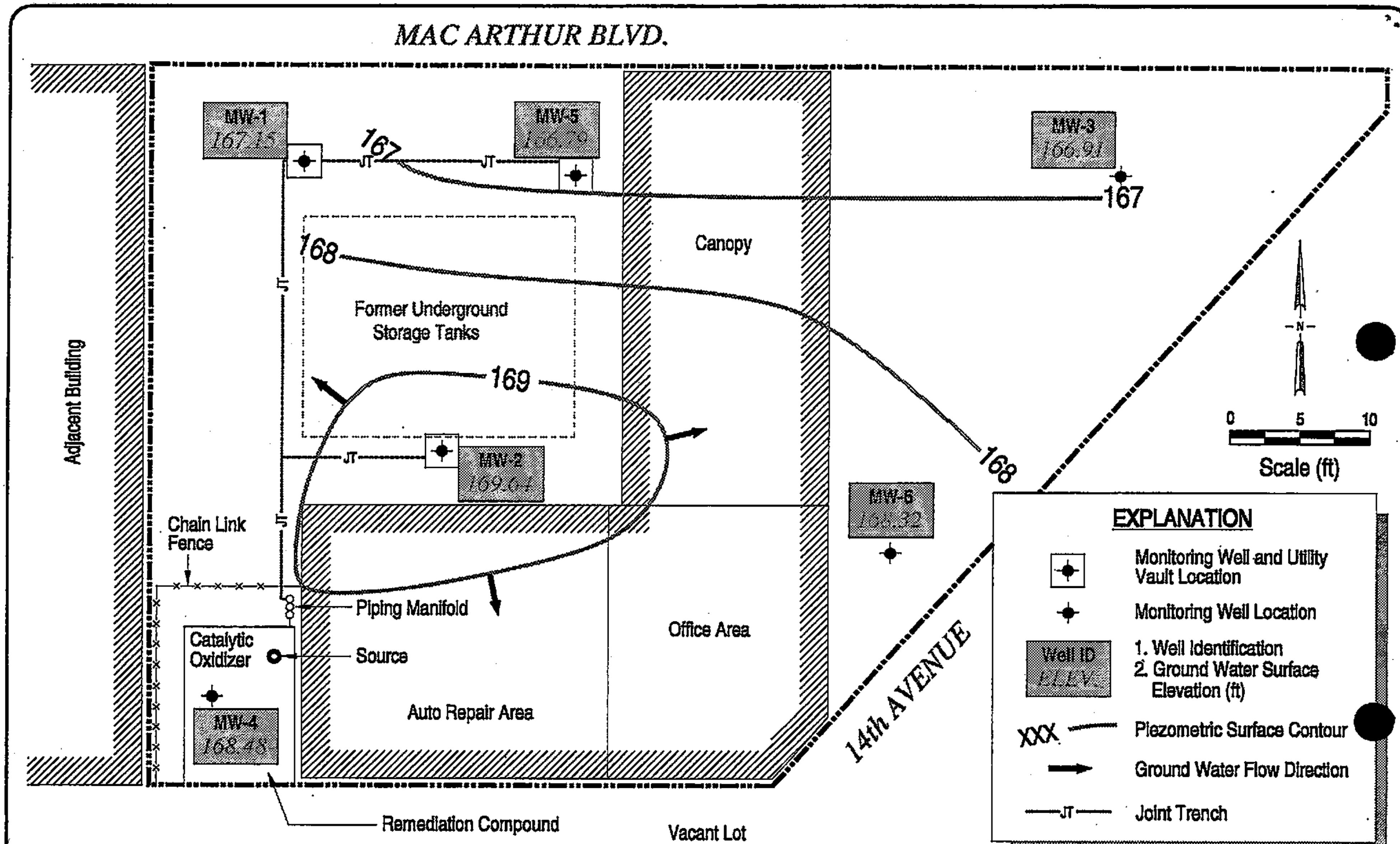
1499 MacAurthur Boulevard

Oakland, California

CAMBRIA

Groundwater Elevation Contour Map

February 17, 2000



NOTE: All points surveyed to an arbitrary datum.

CAMBRIA Environmental Technology, Inc.	Hooshi's Auto Service 1499 MacAurthur Boulevard Oakland, California HSB-2004\OAK-HOOSHI\FIGURES\2CM98-MP.DWG	Ground Water Elevation Countours May 8, 1998	FIGURE **1**

05/26/98