Plume Distortion and Apparent Attenuation Due to Concentration Averaging in Monitoring Wells

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Abstract

Mathematical models that simulate common monitoring well sampling demonstrate the distortions that vertical concentration averaging causes during the mapping and modeling of an idealized, three-dimensional contaminant plume emanating from a simple source of constant solute concentration. The apparent extent of the plume, mapped using simulations of a regular grid of screened monitoring wells, ranged from a worst case of 0% of the original plume area for long screens (4 m) in a low-permeability formation to 90% for short screens (1 m) in a high-permeability formation. When well design and purging procedure were inconsistent among wells, the mapped plume exhibited spurious directional skewing, bifurcation, zones of low concentration, intermittent sources, or multiple sources. Although the study plume was not retarded, calibrating a transport model to the monitoring well data resulted in retardation factors of up to 23. If first-order decay was assumed, the apparent decay constant was found to be as much as 1.8×10^{-7} sec^{-1} ($T_{1/2} = 45$ days). Apparent retardation or decay was inconsistent from well to well, depending on the saturated screen length, the degree of screen desaturation during purging, and the distance from the source. The study indicates that the quantitative assessment of contaminant distributions and transport processes requires discrete vertical sampling in the common situation where concentrations vary sharply with depth, even in the most ideal hydrogeologic environment. If screened monitoring wells are used, screen lengths and placements should be appropriate to the contamination situation being assessed and inherent biases must be considered. Even so, vertical concentration averaging biases and the resulting inconsistencies can result in highly misleading evaluations of ground-water contamination problems.

Introduction

In order to perform ground-water contaminant transport modeling, risk assessment, and effective remediation, dispersion and attenuation parameters must first be determined. Inverse methods have been developed that use concentration distributions to determine dispersion coefficients (Domenico and Robbins, 1985), retardation factors (Mehran et al., 1987), reaction constants (Domenico, 1987), or biodegradation rates (Chiang et al., 1989). The meaningfulness of these parameters depends on the accuracy of plume characterization. Commonly, plumes are mapped using ground-water monitoring wells which, as indicated by recent theoretical and field studies, provide simple to complex concentration averages of the vertical distribution of solute concentrations adjacent to the well screen (Gibs et al., 1993; Gibs and Imbrigiotta, 1990; Martin-Hayden et al., 1991; McLaughlin et al., 1993; Robbins, 1989; Robbins and Martin-Hayden, 1991). The vertically averaged concentrations derived from wells have been shown to depend on such factors as the length and vertical position of the screened interval, the hydraulic properties of the aquifer and of the sand pack around the screen, the vertical distribution of solutes in the vicinity of the well screen, and the method of purging, including the volume of water purged and water levels achieved in the well during purging. Importantly, these factors are seldom consistent from well to well. Hence, contaminant plumes mapped using monitoring wells are distorted due to these effects. Robbins (1989) deduced that the types of distortions that may be observed include apparent plume attenuation, apparent contaminant migration in a direction that is contrary to the hydraulic gradient or the hydrogeology, and unexplainable temporal variations in extent, direction, and rate of migration. Even if the complicating factors are consistent from well to well, concentrations may be underestimated (Robbins and Martin-Hayden, 1991) and, as a result, plumes may seem attenuated. These plume distortions will be

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particularly pronounced where concentrations vary widely with depth because in these situations the biases are sensitive to the vertical position of the well screen relative to the concentration distribution. Consideration of these sensitivities is especially important because recent detailed studies of vertical concentration distributions have indicated that vertical spreading is often limited and that concentrations are often highly variable with depth (Garabedian et al., 1987; Gibs et al., 1993; Martin-Hayden et al., 1991; and Sudicky et al., 1983).

The objectives of this study were to examine how vertical concentration averaging may distort our understanding of a plume and also influence the magnitude of retardation and first-order decay factors (such as biodegradation rate constants) calculated using the biased concentrations. The study entailed the mathematical generation of an idealized three-dimensional contaminant plume with attributes that are common to many field situations. In order to simulate the sampling of this contaminant plume, monitoring well sampling models were applied to the concentration distribution to yield monitoring well sample concentrations. These sample concentrations were then used to create plume maps and to calibrate various transport models. The resulting maps and calibrations demonstrate the plume distortions that may arise from sampling screened monitoring wells and the resulting miscalibration of attenuation parameters. The contaminant situation examined is the common case where concentrations are high near the water table and drop off sharply with depth. This situation was chosen not only for its typicality but also to demonstrate the range of possible distortions (including worst-case scenarios) if the concentration distribution is sampled using screened monitoring wells of common lengths and purging practices. For other situations where the concentrations vary widely with depth, the specific outcomes of these evaluations would be different and possibly less severe. The overall conclusions, however, would be the same.

Methodology

An idealized three-dimensional concentration distribution (referred to as the study plume) was mathematically generated using the three-dimensional extended pulse model of Domenico and Robbins (1985) for the case of downward solute spreading from the water table. This model assumes that uniform flow parallel to the water table transports solute from a continuous source of constant solute concentration with no retardation or degradation. The Domenico and Robbins model incorporates the truncated version of the one-dimensional continuous source model of Ogata and Banks (1961), rather than the full expression, in order to simplify the expression of their model. This truncation leads to large errors (between 5% and 25%) in the situation modeled in this study. To eliminate these errors, the models used here incorporate the full Ogata and Banks term [f_x in equation (1)] rather than the truncated version used by Domenico and Robbins. With this modification, the equation describing the three-dimensional concentration distribution becomes

$$C(x, y, z, t) = \frac{C_o}{8} \cdot f_x \cdot f_y \cdot f_z \qquad (1)$$

where,

$$f_x = \text{erfc}\left(\frac{(x - vt)}{2(\alpha_x vt)^{1/2}}\right) + \exp\left(\frac{x}{\alpha_x}\right) \cdot \text{erfc}\left(\frac{(x + vt)}{2(\alpha_x vt)^{1/2}}\right);$$

(full Ogata and Banks term)

$$f_y = \text{erf}\left(\frac{(y + Y/2)}{2(\alpha_y x)^{1/2}}\right) - \text{erf}\left(\frac{(y - Y/2)}{2(\alpha_y x)^{1/2}}\right);$$

$$f_z = \text{erf}\left(\frac{(z + Z)}{2(\alpha_z x)^{1/2}}\right) - \text{erf}\left(\frac{(z - Z)}{2(\alpha_z x)^{1/2}}\right);$$

x is the horizontal coordinate parallel to the direction of flow with the origin at the source (i.e., x is the longitudinal distance from the source, positive in the direction of flow); y is the horizontal coordinate in the direction perpendicular to the direction of flow with the origin at the center of the source; z is the vertical coordinate with $z = 0$ at the water table and positive downward; C_o is the source concentration; Y and Z are the width and height of the source, respectively; α_x, α_y, and α_z are the dispersivities of the solute being modeled; t is the time since onset of contamination; and v is the average linear velocity (i.e., specific discharge divided by effective porosity). Table 1 summarizes the parameters used to generate the concentration distribution shown in Figure 1 (the study plume). Implicit in these parameters, which are within the range of those commonly reported, is a continuous source of constant solute concentration (i.e., constant leaching rate) that originates from just below the water table. Also note that the transport of this solute is taken as conservative (i.e., experiences no retardation or degradation).

Monitoring well sampling of the three-dimensional concentration distribution was simulated using models developed by Robbins and Martin-Hayden (1991) which calculate the average or composite average (weighted average) of the vertical concentration distribution. In the simplest situation the concentration in the monitoring well following purging will be the average of the vertical concentration distribution between the top and the bottom of the screen. This occurs if the ground-water flow to the well is predominantly horizontal and uniform along the well screen during purging, as might be the case for a well within a formation with a high hydraulic conductivity which is uniform with depth and where the screen remains fully saturated. For a continuous vertical concentration distribution adjacent to the screen, the average of the vertical concentration distribution, $\bar{C}_z$, is:

$$\bar{C}_z = \frac{1}{b_1 - b_2} \int_{b_2}^{b_1} C(z)\, dz \qquad (2)$$

where b_1 and b_2 are the z coordinates of the bottom and the top

Table 1. Model Parameters Used to Calculate the Three-Dimensional Contaminant Concentration Distribution

Model parameter	*Symbol, value*
Source width	$Y = 10$ m
Source height	$Z = 0.25$ m
Source concentration	$C_o = 1000\ \mu g/l$
Longitudinal dispersivity	$\alpha_x = 5$ m
Transverse dispersivities:	
Horizontal	$\alpha_y = 0.5$ m
Vertical	$\alpha_z = 0.01$ m
Seepage velocity	$v = 1 \cdot 10^{-7}$ m/s
Time	$t = 2.523 \cdot 10^9$ s (8 yrs)

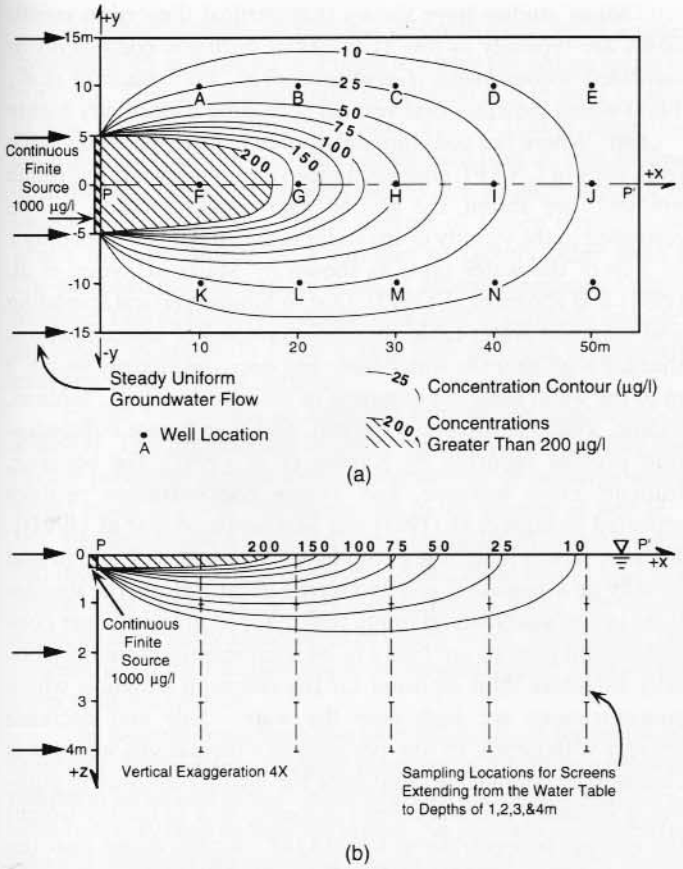

Fig. 1. Study plume: Simulated three-dimensional concentration distribution (a) map view of concentrations at the water table, and (b) cross section along centerline.

of the saturated portion of the screen, and C(z) is concentration as a function of the vertical coordinate. In this study C(z) is described by equation (1).

In a more complicated situation, where the well screen is desaturated during purging, Robbins and Martin-Hayden (1991) demonstrated that the concentration in the monitoring well following purging will be a composite average of the vertical concentration distribution adjacent to the saturated portion of the screen. Composite averaging is the result of screen desaturation and water level recovery as each incremental bailer volume of water is removed during purging. This screen desaturation leads to preferential flow through the saturated portion of the screen and weights the average concentration toward groundwater concentrations near the bottom of the well screen. For the case of a continuous concentration distribution, C(z), the concentration within a purged monitoring well when composite averaging has occurred is:

$$\overline{\overline{C}}_z = \frac{1}{(b_2 - b_3)} \int_{b_3}^{b_2} \left(\frac{1}{(b - b_1)} \int_{b_1}^{b} C(z)\, dz \right) db \quad (3)$$

where $\overline{\overline{C}}_z$ is the composite average of the vertical concentration distribution, b_3 is the water level immediately after each bailer volume is removed during purging, and b_1 is the bottom of the screen as defined for equation (2). This model assumes that the water level is allowed to fully recover following the removal of each bailer volume during purging and that no water enters the desaturated portion of the screen as the well resaturates. Equation (3) demonstrates that the degree to which the average is weighted toward the concentrations at the bottom of the screen depends on the percentage of screen desaturation that occurs during purging. While this model may overestimate the weighting in some cases because it does not account for the possibility of water entering the screen from above the water level (i.e., across a seepage face), the analysis will indicate the possible range of variability that might be expected. The percentage of screen desaturation is calculated as $100*(b_2 - b_3)/(b_2 - b_1)$ and, because equation (3) is undefined when $b_3 = b_1$, full (100%) desaturation is taken as $100*(b_2 - b_3)/(b_2 - b_1) = 99.99\%$.

Concentration averaging simulations were performed for each well (locations A through O) on Figure 1 using equation (2) for the case of 0% screen desaturation and equation (3) for the case of screen desaturation greater than 0%. The integrations of C(z) within equations (2) and (3) were performed by numerically integrating equation (1). To represent a range of typical monitoring well constructions, the well screens were assumed to extend from the water table to depths of 1, 2, 3, or 4 m below the water table. The extent of screen desaturation during purging was assumed to be 0 (fully saturated), 25, 50, 75, or 100% (fully desaturated). Average concentrations calculated were then used to examine the following issues. To what extent might concentration averaging influence estimates of the size of a plume (i.e., area contaminated above an action level)? Because screen lengths and water levels during purging are often inconsistent between wells, to what extent might this inconsistency influence the apparent migration direction of a plume, or influence the determination of the location, the concentration, and the condition of the source? To what extent might concentration averaging influence the apparent rate of plume migration or the extent to which processes are acting to attenuate a plume?

Simulation Results and Discussion

Apparent Area of Contamination

Figure 2 shows four cases of apparent plume area mapped using data from wells with the same screen length throughout the well field and 0% screen desaturation, given the concentration distribution of Figure 1. These cases are shown on the figure as four 10 µg/l contour lines mapped using a field of 1 m well screens, a field of 2 m well screens, a field of 3 m well screens, and

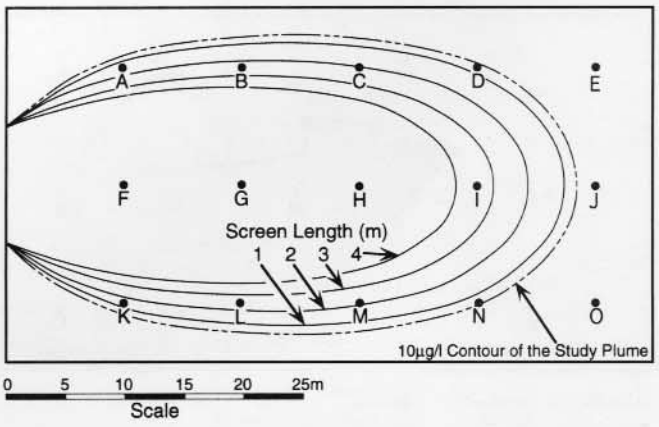

Fig. 2. Apparent areas of contamination (10 µg/l contours) mapped using screen lengths of either 1, 2, 3, or 4 m with 0% screen desaturation during purging.

a field of 4 m well screens whose concentrations were linearly interpolated and hand smoothed. The area of contamination is taken as the area encompassed by this 10 μg/l contour line (i.e., concentrations of more than 1% of the C_o of 1000 μg/l which may be some maximum allowable concentration or detection limit of the solute being investigated). As shown on Figure 2, the apparent area of contamination decreases with increasing screen length. For this simulation the apparent area of contamination varied by a factor of two from 530 m^2 for the 4 m screen to 930 m^2 for the 1 m screen. The actual area of contamination in Figure 1 was 1000 m^2 based on the 10 μg/l contour line.

Figures 3 and 4 show the effect of various degrees of screen desaturation during purging on the apparent area of contamination in the cases of 1 m and 4 m screen length, respectively. For a 1 m screen (Figure 3), screen desaturation will cause further dilution of the monitoring well concentrations and further reduction of the apparent area of contamination. Here the plume area diminished from 930 m^2 for the fully saturated case to 809 m^2 for the case of full screen desaturation. This effect becomes even more dramatic as the screen length is increased to 4 m (Figure 4), considerably greater than the vertical extent of contamination. For a screen length of 4 m the area of contamination diminishes from 530 to 0 m^2 as composite averaging has diluted the entire plume to a concentration below the 10 μg/l limit.

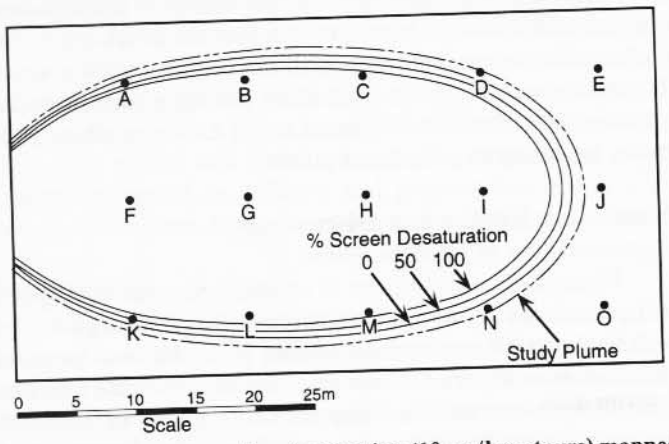

Fig. 3. Apparent areas of contamination (10 μg/l contours) mapped using 1 m screen lengths with either 0, 50, or 100% screen desaturation during purging.

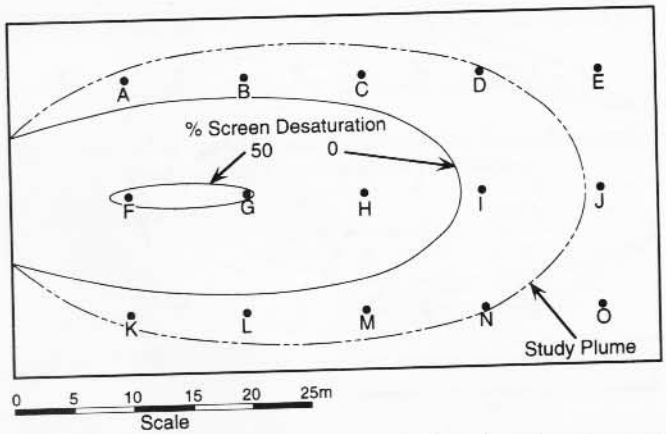

Fig. 4. Apparent areas of contamination (10 μg/l contours) mapped using 4 m screen lengths with either 0, 50, or 100% screen desaturation during purging.

Many studies have shown that vertical dispersion coefficients are typically as low as values of diffusion coefficients in saturated porous media (Garabedian et al., 1987; Sudicky et al., 1983) which indicates that vertical spreading is normally highly limited. Where the contaminant source is near the water table (e.g., sites of LNAPL contamination) and downward hydraulic gradients are absent, the ground-water contamination will be restricted to the vicinity of the water table, possibly within only 1 or 2 m of the water table as shown by Martin-Hayden et al. (1991) and Raven et al. (1992). Due to limited vertical spreading and common water-table sources, examples of concentrations that are high near the water table and decrease sharply within 2 m of the water table are common in the literature [e.g., toluene, xylene, xylidine, dissolved oxygen, and manganese concentration profiles reported by Kaplan et al. (1991); the benzene, toluene, ethyl benzene, and xylene concentration profiles reported by Gibs et al. (1993) and Martin-Hayden et al. (1991)]. Because screen lengths are commonly longer than 2 or 3 m and usually no attention is given to screen desaturation, the simulations in this study would imply that mapped ground-water contamination plumes are likely to be significantly more horizontally extensive than reported for the common situation where concentrations are high near the water table and decrease sharply with depth. In low-permeability formations where it is common to completely desaturate the well screen during purging, a situation which has been demonstrated to further weight the average concentrations toward the concentrations near the bottom of the well (Martin-Hayden et al., 1991), plume sizes are especially likely to be underestimated.

Plume Distortion

In the preceding section each simulated well field included monitoring wells with uniform lengths and consistent degrees of screen desaturation. In an actual field situation screens are seldom the same length (due to inconsistencies in construction and geologic conditions) and typically no attention is given to the degree of screen desaturation during purging. Different degrees of screen desaturation may result if the screens and sand packs have become clogged to different degrees, if the sand packs are not the same, or if the wells are purged at different rates. These inconstancies will distort the shape of the plume. To examine the types of distortions that may arise, the simulations shown in Figures 5 through 8 were generated using different combinations of screen lengths and degrees of screen desaturation. It should be realized that there are an infinite number of possible plume distortions. The figures were chosen to illustrate different categories of distortions that may arise, relative to the original plume shown in Figure 1.

Figure 5 shows that a mapped plume based on inconsistent screen lengths and purging practices might appear to trend in a direction that is skewed relative to the hydraulic gradient. This might be misinterpreted as resulting from anisotropy of the hydraulic conductivity that would cause the ground water to flow in a direction different from the hydraulic gradient. Alternatively, this might be misinterpreted as being due to temporal variations in the hydraulic gradient. The plume also appears to be emanating from a different place than that of the true source of the study plume. Such a situation could clearly lead to unwarranted investigation to delineate a fictitious source or lead to unfounded regulatory or legal actions.

Another example of possible misinterpretation of aquifer hydrogeology is depicted by Figure 6, where the plume seems to bifurcate. This might lead to the conclusion that there is an area of low hydraulic conductivity near well G that is impeding contaminant transport so that contaminant flow divides around it. However, the lower concentrations are actually a result of longer well screens and higher degrees of screen desaturation which may be caused by a number of factors other than a low-permeability formation, such as screen clogging or rapid purging. This situation emphasizes the need to carefully consider geological information from boring logs in order to properly interpret contaminant patterns.

Figure 7 illustrates an apparent zone of low concentration at well H. This plume shape could be misinterpreted as being due to such processes as enhanced biodegradation or recharge which would result in lower concentrations in the vicinity of well H.

Figure 8 shows a situation where inconsistent screen lengths and inconsistent degrees of screen desaturation would lead to a plume with two areas of high concentration. Here, the source might be incorrectly assumed to be intermittent. Alternately, the concentration distribution might be misinterpreted as resulting from multiple sources.

The illustrations above also have implications for temporal sampling. Inconsistent purging practices can lead to apparent temporal variations in well concentrations and plume configurations that may then be misinterpreted. An increase in concentration at a well between sampling rounds brought about by inconsistent purging might be misinterpreted as an increase in source strength. A decrease in concentration might be misinterpreted as being due to the effectiveness of remediation. The magnitudes of these effects are illustrated in Figures 3(a) and 4(a). In Figure 3(a), the concentration at well F can be made to vary by a factor of two depending on how it is purged. For a longer screen length, as illustrated in Figure 4(a), the concentration at well F can vary by an order of magnitude. These concentration variations illustrate the need for consistency in the manner in which wells are purged (i.e., water levels achieved during purging) in order to discern natural temporal trends in concentration. Even then, it

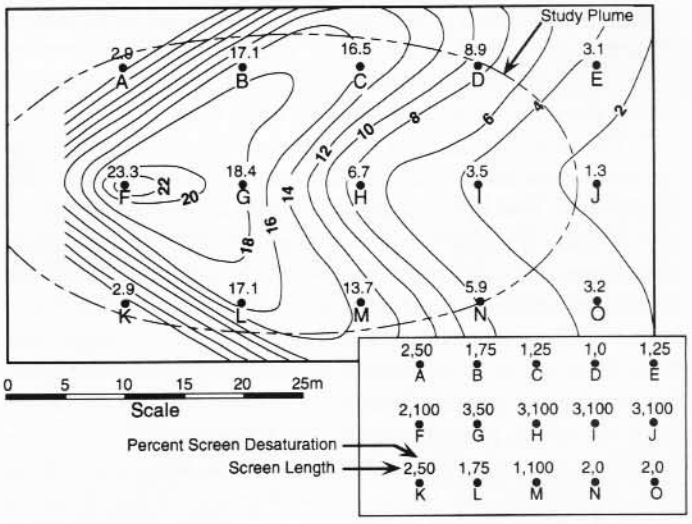

Fig. 6. Concentration contours showing apparent plume bifurcation.

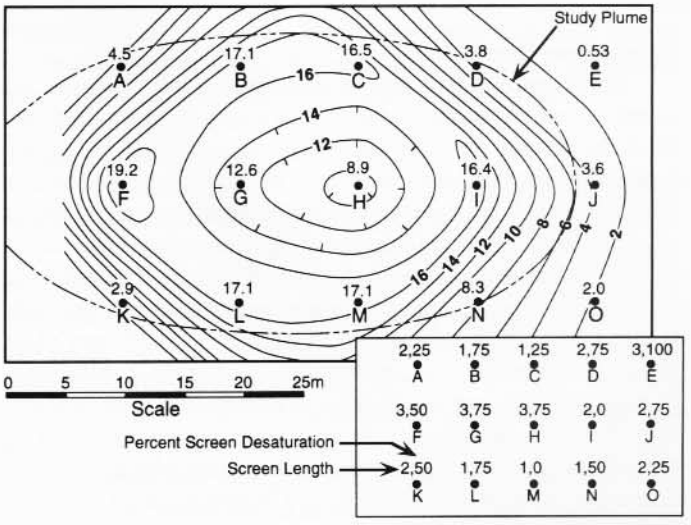

Fig. 7. Concentration contours showing apparent zone of low concentration.

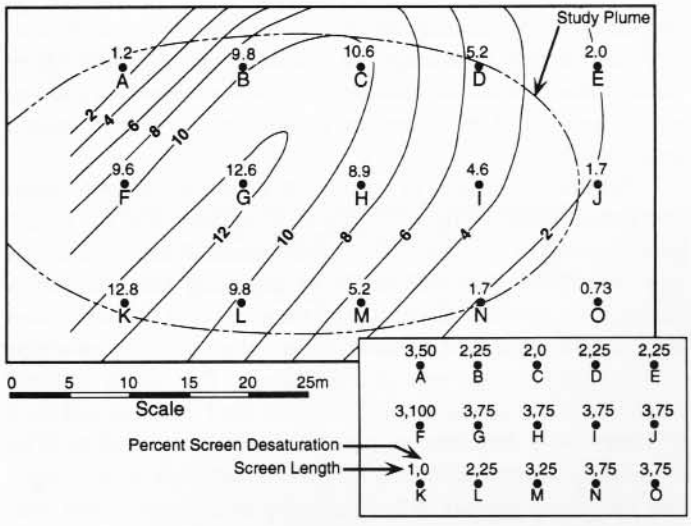

Fig. 5. Concentration contours showing misleading apparent plume migration direction and source location relative to the actual orientation of the study plume.

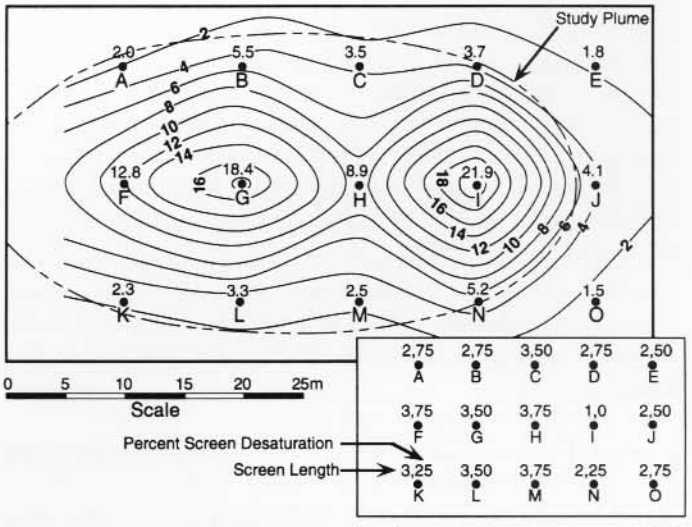

Fig. 8. Concentration contours showing apparent zones of high concentration.

must be realized that concentrations observed will be a function of water-table levels which may change between sampling rounds. Hence, the water levels in the well must be considered during the interpretation of concentration data.

The above simulations assumed simple hydrology and contaminant conditions and yet illustrate the complexities that might result. Had this contaminant situation involved complexities in mass transport, hydrogeology, chemistry, and source conditions, artificial irregularities would be superimposed on the natural complexities making it difficult or even impossible to correctly characterize the contaminant situation. Although these distortions might be minimized by installing wells with consistent designs and purging them in a consistent manner, the distortions will still be present because they are dependent on concentration averaging over a finite screen interval, water levels at the time of sampling, and the specific stratigraphy in the vicinity of the well. With respect to the latter, average concentrations obtained from well samples will be weighted toward more permeable and thicker strata adjacent to the well screen (Cohen and Rabold, 1988; Gibs et al., 1993; Reilly and Gibs, 1993; Robbins and Martin-Hayden, 1991).

Attenuation Parameters

Using monitoring well concentrations to delineate the extent of contamination, as shown in Figures 2 through 4, will result in plumes that are apparently attenuated (i.e., exhibit concentrations that are lower than would be expected with a conservative solute). Apparent plume attenuation may be recognized through model calibration or by comparing the extent of the mapped plume relative to the advective front. The advective front is defined as $v \cdot t$ and is calculated using measured hydraulic gradients, hydraulic conductivities, porosities, and Darcy's law. In order to simulate the apparent plume using a contaminant transport model, one would have to incorporate retardation or degradation to account for the apparent plume attenuation. In the following sections we examine the magnitudes of apparent retardation or first-order degradation factors required to achieve model calibration.

Apparent Retardation Factor

The retardation factor is defined as $R_f = v/v_i$, where v is the average linear velocity, and v_i is the attenuated contaminant velocity. Robbins (1989) noted that two-dimensional models are commonly used to calculate the R_f of observed contaminant plumes. In this case we have used the two-dimensional form of the continuous pulse model of Domenico and Robbins (1985) modified to incorporate the full Ogata and Banks term, as was done for equation (1), and modified to incorporate retardation in the same way Domenico and Schwartz (1990) modified the 3-D extended pulse model. The two-dimensional concentration distribution, C_{2D}, is given by

$$C_{2D}(x, y, t) = \frac{C_o}{4} \cdot g_x \cdot f_y \qquad (4)$$

where,

$$g_x = \operatorname{erfc}\left(\frac{(x - vt/R_f)}{2(\alpha_x vt/R_f)^{1/2}} \right) + \exp\left(\frac{x}{\alpha_x} \right) \cdot \operatorname{erfc}\left(\frac{(x + vt/R_f)}{2(\alpha_x vt/R_f)^{1/2}} \right),$$

Table 2. Variation in Apparent Retardation Factor with Distance from Source

	Apparent retardation factor	
Distance from source	*1 m Screen 0% Screen Desaturation*	*4 m Screen 100% Screen Desaturation*
10 m	7.10	23.50
20	2.94	6.73
30	1.97	3.55
40	1.58	2.41
50	1.38	1.88
Average:	2.99	7.61

f_y and the other parameters are described for equation (1). The transport parameters used were those shown in Table 1. This model, however, does not require a vertical source dimension or a vertical dispersivity. This analysis demonstrates how the problems that result from calibrating a two-dimensional model to a three-dimensional contaminant distribution will be exacerbated by using screened monitoring wells of inappropriate lengths and purging practices. Similar problems will result if screens of inappropriate lengths and vertical positions are used to calibrate three-dimensional contaminant transport models.

Apparent retardation factors were first calculated at each well (A through O) by assuming v was known and adjusting the retardation factor until the model concentration matched the monitoring well concentration. Table 2 shows the retardation factors calculated at the wells as a function of longitudinal distance from the source (x coordinate) for the two extremes of 1 m screens with 0% desaturation and 4 m screens with 100% desaturation. The values for the other amounts of screen desaturation and for the 2 m and 3 m screens will fall between these limits. The table shows that as a result of concentration averaging there is apparently no unique retardation factor for the well field. The apparent retardation factor will increase with increasing screen length and degree of desaturation, and will decrease with longitudinal distance from source. Such a result might be interpreted as being due to attenuation processes other than retardation. Alternatively, an investigator might assume that the variability of the retardation factor might be due to analytical error and decide to average the results to get a representative value. Table 3 shows the average retardation factors for the 1 m screen and the 4 m screen as a function of percent screen desaturation.

Another approach to estimating a representative retardation factor would be to perform a least-squares fit of the model [equation (4)] to the observed concentrations. A least-squares fit was accomplished in this study by adjusting the retardation factor of the model until the sum of the squares of the differences between the measured concentration values and the modeled concentration values were at a minimum. The results of these calculations are also shown in Table 3 for the 1 m screen and the 4 m screen as a function of percent screen desaturation. The retardation factors estimated by inverse methods would imply that the plume appears to be moving up to 25 times slower than the true rate of migration of the study plume. This would lead to the serious underestimation of the risk involved in a contamination situation.

Apparent Decay Constant

If the attenuation mechanism was assumed to be first-order decay, the two-dimensional transport model used to simulate the contaminant distribution may be modified to account for first-order degradation (Domenico and Schwartz, 1990).

$$C_{2D}(x, y, t) = \frac{C_o}{4} \cdot h_x \cdot f_y \qquad (5)$$

where,

$$h_x = \exp\left(\frac{x[1 - (1 + 4\lambda\alpha_x/v)^{1/2}]}{2\alpha_x} \right) \cdot \operatorname{erfc}\left(\frac{x - vt(1 + 4\lambda\alpha_x/v)^{1/2}}{2(\alpha_x vt)^{1/2}} \right) + \exp\left(\frac{x[1 + (1 + 4\lambda\alpha_x/v)^{1/2}]}{2\alpha_x} \right) \cdot \operatorname{erfc}\left(\frac{x + vt(1 + 4\lambda\alpha_x/v)^{1/2}}{2(\alpha_x vt)^{1/2}} \right),$$

λ is the decay constant in s^{-1}, and f_y and the other parameters are described for equation (1). The calibration process used to create Tables 2 and 3 was repeated for the decay constant. Table 4 shows the decay constants needed at the wells as a function of longitudinal distance from the source for the two extremes of 1 m screens with 0% desaturation and 4 m screens with 100% desaturation. The table shows that there is apparently no common decay constant for the well field. The decay constants will vary depending on screen length, degree of screen desaturation, and longitudinal distance from the source. Table 5 shows the average retardation factors for the 1 m screen and the 4 m screen as a function of screen desaturation. Table 5 also shows the results of the least-squares fit of the model to the monitoring well data for the 1 m screen and the 4 m screen as a function of screen desaturation. These tables indicate that the apparent decay factor might be assessed to be as much as $1.8 \cdot 10^{-7}$ s^{-1} which is equivalent to a half-life of 45 days. These values are in the range of those reported for the aerobic biodegradation of aromatic hydrocarbons (e.g., Chiang et al., 1989) based on data from screened monitoring wells. Hence, apparent decay can lead to erroneous overestimation of degradation processes, and consequently to underestimation of plume extent and risk.

Table 3. Variation in Average Apparent Retardation Factor and Least-Squares Fit Retardation Factor with % Screen Desaturation

	Average apparent retardation factor		*Least-squares fit retardation factor*	
Screen desaturation	*1 m Screen*	*4 m Screen*	*1 m Screen*	*4 m Screen*
0%	2.99	4.72	7.01	13.0
50	3.26	6.66	8.09	20.3
100	3.63	7.61	9.62	23.5

Table 4. Variation in Apparent Decay Constant with Distance from Source

	Apparent decay constant (s^{-1})	
Distance from source	*1 m Screen 0% Screen Desaturation*	*4 m Screen 100% Screen Desaturation*
10 m	$2.61 \cdot 10^{-8}$	$1.84 \cdot 10^{-7}$
20	$1.29 \cdot 10^{-8}$	$5.60 \cdot 10^{-8}$
30	$9.98 \cdot 10^{-9}$	$3.23 \cdot 10^{-8}$
40	$8.93 \cdot 10^{-9}$	$2.43 \cdot 10^{-8}$
50	$8.52 \cdot 10^{-9}$	$2.07 \cdot 10^{-8}$
Average:	$1.33 \cdot 10^{-8}$	$6.33 \cdot 10^{-8}$

Table 5. Variation in Least-Squares Fit Decay Constant and Average Apparent Decay Constant with % Screen Desaturation

	Average apparent decay constant (s^{-1})		*Least-squares fit decay constant (s^{-1})*	
Screen desaturation	*1 m Screen*	*4 m Screen*	*1 m Screen*	*4 m Screen*
0%	$1.33 \cdot 10^{-8}$	$2.85 \cdot 10^{-8}$	$2.03 \cdot 10^{-8}$	$6.55 \cdot 10^{-8}$
50	$1.51 \cdot 10^{-8}$	$5.03 \cdot 10^{-8}$	$2.80 \cdot 10^{-8}$	$1.45 \cdot 10^{-7}$
100	$1.79 \cdot 10^{-8}$	$6.33 \cdot 10^{-8}$	$3.86 \cdot 10^{-8}$	$1.82 \cdot 10^{-7}$

Conclusions

This study indicates that the quantitative assessment of contaminant distribution and transport processes requires discrete vertical sampling, even in the most ideal hydrogeologic environment. The inherent bias in water-quality data obtained from monitoring wells, brought about by vertical concentration averaging, can lead to highly misleading evaluations of groundwater contamination problems. Although we have provided advice on the qualitative assessment of vertical concentration averaging bias, without knowledge of the vertical distribution of contamination and hydraulic conductivity it is not possible to quantitatively assess bias in monitoring well data. The results of this study would also imply that the use of screened monitoring wells to evaluate contamination problems should be reassessed. At the very least, where conditions require the use of screened monitoring wells, they should be: constructed with screens of consistent length that are as short as possible; placed at depths which are appropriate to the geology, source conditions, and contaminant distribution and transport; and sampled in a consistent manner with as little screen desaturation as possible. Field screening methods and on-site hydrogeological characterization during drilling may be used to provide the information required for designing the shortest possible screen lengths and appropriate vertical placements of the screens relative to the concentration distributions and hydrogeology at each specific site. Minimization of the bias due to concentration averaging effects will improve the validity of transport model predictions, and ultimately improve the effectiveness of remediation design based on both the mapped contaminant distributions and the transport modeling.

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