Hydraulic Characterization and Design of a Full-Scale Biocurtain

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Abstract

This paper describes the design and hydraulic characterization of a cost-effective biocurtain that is currently being used to remove carbon tetrachloride from an aquifer in Schoolcraft, Michigan. Novel aspects of the design are the use of closely spaced wells to recirculate solutes through a biocurtain, well screens spanning the vertical extent of contamination, and a semipassive mode of operation, with only six hours of low-level pumping per week. This design was developed by coupling flow and transport simulations with a cost optimization algorithm, based on initial hydraulic conductivity data and system design constraints from a previous pilot-scale experiment adjacent to the current site. The hydraulic conductivity of the site was characterized using permeameter analysis on more than 200 samples from continuous well cores that were collected during well installation. The subset of available conductivity data was used to predict tracer transport through the biocurtain during system operation. Observed tracer concentration arrival histories during initial system operation confirmed model predictions. Modeling also established that closely spaced wells operated for brief periods each week could effectively deliver the agents needed for remediation across the biocurtain. This was confirmed during long-term operation of the system, which has resulted in highly efficient contamination degradation. The delivery well design methodology is expected to be broadly applicable at other sites where flow can be recirculated between a series of delivery wells.

Introduction

Ground water contamination with volatile organic compounds is a significant national and international problem. Water containing these contaminants is typically pumped from contaminated aquifers and treated by air stripping or sorption onto activated carbon. These methods are costly, do not destroy the contaminants, may require pumping and disposal of large volumes of water, and do not effectively remove contaminants sorbed to the aquifer material. Accordingly, there has been a great deal of interest in alternative treatment strategies, such as enhanced in situ remediation. Over the last decade, bioremediation has become increasingly common and accepted. The three basic bioremediation methods are natural attenuation, biostimulation, and bioaugmentation. Natural attenuation is now widely accepted for management of some hydrocarbon plumes. Biostimulation is also practiced in cases where the addition of oxygen in a contaminant source area stimulates microbial activity, and thus enhances remediation efficiency. For chlorinated organics, natural attenuation may be insufficient, and electron donor addition is often needed to accelerate reductive dechlorination or to stimulate cometabolic activities. In some cases, indigenous bacteria may be unable to mediate the desired transformation (Dybas et al. 1995, 1998; Ellis et al. 1999), thus microorganisms with the desired traits will need to be added (bioaugmentation). For both biostimulation and bioaugmentation, a critical issue is chemical delivery within the treatment zone to maintain microbial activity; and for bioaugmentation, an additional issue is organism delivery.

Delivery can be a significant problem because the transport of contaminants and the agents needed for remediation are profoundly affected by the heterogeneous properties of aquifer sediments (Dybas et al. 1998) and by the complicated temporal and spatial nature of contaminant releases. The influence of heterogeneity on solute transport has been demonstrated in various geologic settings. These include the fairly homogeneous glacial deposits of Borden Ontario (Freyberg 1986), the glacial outwash of the Cape Cod Aquifer in Massachusetts (Hess et al. 1992; LeBlanc et al. 1991; Garabedian et al. 1991), and the heterogeneous alluvial deposits of the MacroDispersion Experiment site (MADE) in Mississippi (Adams and Gelhar 1992; Boggs et al. 1992; Rehfeldt et al. 1992). Aquifer characterization is thus a critical step in the design of enhanced in situ remediation methods in most geologic settings.

One approach to enhance delivery in a heterogeneous shallow aquifer is to emplace a permeable wall by pounding sheet piling into the subsurface, dewatering a region, backfilling with filter sand, and removing the sheet piling (Devlin and Barker 1996). However, this approach is costly and is not feasible in deep aquifers. As we will demonstrate, it is possible to design a system around the characterized heterogeneity rather than overcoming the heterogeneity.

Several pilot- and full-scale studies have demonstrated effective degradation of chlorinated organics (Semprini et al. 1992, 1994; Ellis et al. 1999). Recently, McCarty et al. (1998) demonstrated full-scale cometabolic degradation of trichloroethlylene (TCE) at the Edwards Air Force Base, California. Toluene and oxygen were added to stimulate indigenous toluene degrading microflora capable of cometabolically degrading TCE. Hydrogen peroxide was added as the oxygen supply during most of this experiment to reduce the likelihood of biofouling. This landmark

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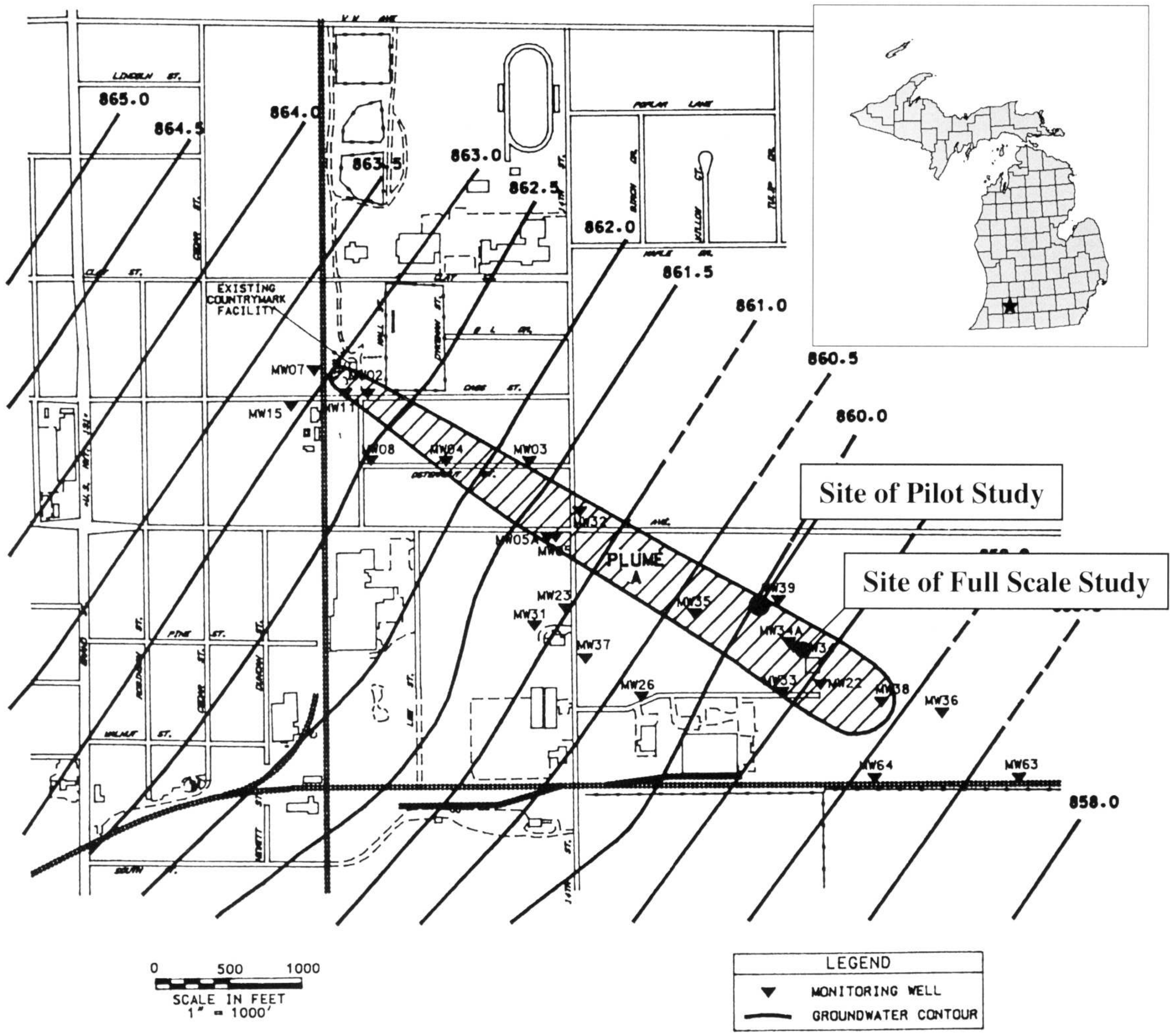

Figure 1. Map of the Schoolcraft, Michigan, region showing plume A and water table contours in feet, modified from Mayotte et al. (1996). The locations of our current bioaugmentation system and the pilot scale study site are noted.

study demonstrated highly efficient removal of TCE (99.98%), using paired wells to create the desired flow regime and to periodically inject toluene. However, generalization of the design used at this site is complicated by the fact that it relied upon a rather unique hydrogeological setting, with flow passing between two aquifers. Moreover, this study did not need to confront the issue of bacterial transport because indigenous microflora mediated the transformation.

In this paper, we present the design of a delivery well system for bioaugmentation, which should have broad applicability to confined or unconfined heterogeneous aquifers. The design appears to be suitable for introduction of all of the agents required for bioremediation, including bacteria. The underlying concept is that a plume of contaminated ground water is allowed to flow passively (i.e., under the influence of the natural hydraulic gradient) into a treatment zone or "biocurtain," where bacteria attached to the sediments degrade contaminants. Biodegradation occurs after stimulation of the attached bacteria by intermittent flushing of the treatment zone with chemicals needed to support their activity (acetate, base, and phosphorus, in this case). Inoculation and long-term chemical delivery is accomplished using a row of closely spaced delivery wells (the delivery well "gallery") screened across the vertical extent of contamination. Ground water is periodically extracted from alternating delivery wells, amended with the chemicals needed to support bioremediation, and injected into adjacent wells across the delivery well gallery. The intermittent feeding schedule maintains adequate biomass to efficiently remove contaminants without significant reduction in aquifer conductivity. For bioaugmentation, the delivery wells are used to both inoculate the biocurtain with the desired organisms and to maintain conditions that allow the added organisms to compete with indigenous microflora. The resulting biocurtain is a thin region of biological activity perpendicular to the natural ground water flow direction. In our work, we did not attempt to remediate the entire plume; however, we did create a full-scale biocurtain that treats a significant fraction of the plume. To our knowledge, this is the first presentation of a full-scale biocurtain design that has efficiently remediated contaminated ground water for an extended time period (approximately two years). Although our efforts have focused on bioremediation of a specific aquifer in Michigan, the developed methods should be applicable to a wide range of aquifers and contaminants.

Schoolcraft Site Background

The problem we addressed is remediation of a chlorinated solvent plume in an unconfined aquifer where the contaminant is spread over a large depth range. The contaminant is carbon tetrachloride (CT), a suspected human carcinogen that causes acute liver toxicity in animals and contributes to ozone depletion (Sittig 1985; United Nations Environment Programme 1994). CT releases have occurred because of its past use as a fumigant, dry cleaning agent, fire retardant, and solvent for plutonium refining. The site we examined was Schoolcraft Plume A (Figure 1), a CT-contaminated region of an unconfined aquifer in southwestern Michigan. Schoolcraft Plume A is about 1.2 km long, 90 m wide, and extends from roughly 8 to 26 m below ground surface (bgs). Ground water at the site also contains significant nitrate contamination (50 to 80 ppm), presumably from past agricultural practices. The aquifer is composed of a sequence of approximately 27 m of glaciofluvial sediments overlying a regional clay layer, with the water table approximately 4.5 m bgs.

Pilot-Scale Experiment

Previously, we conducted a pilot-scale experiment in Schoolcraft Plume A to evaluate the potential for bioremediation of carbon tetrachloride (CT) by bioaugmentation with *Pseudomonas stutzeri* KC (hereafter referred to as strain KC) (Dybas et al. 1998). Under denitrifying conditions, this organism degrades CT without producing chloroform (CF) (Criddle et al. 1990; Lewis and Crawford 1999). CT transformation and growth of strain KC require the presence of nitrate as an electron acceptor and an electron donor (such as acetate), sufficient nutrients, and a moderately alkaline pH (7.8 to 8.3). Schoolcraft ground water contains CT (up to approx-

Table 1
Major Design Objectives and How They Were Satisfied

Objectives	Strategies Chosen	Rationale for Strategies	Data Used to Develop Strategies	Reference
1. Develop system to meet regulatory limits.	Prior to inoculation, introduce weekly "pulses" of base-amended ground water adjusted to pH 8.2.	pH 8.2 creates iron-limiting conditions needed for strain KC survival and CT degradation.	Results of bench- and pilot-scale studies	Dybas et al. (1995, 1998) Tatara et al. (1993, 1995)
	Inoculate with sufficient strain KC, then supply weekly pulses of acetate (100 mg/L, later reduced to 50), base (pH 8.2), + phosphorus (10 mg/L).	Stoichiometric removal of nitrate supports long-term removal of 100 µg/L CT to below regulatory limits. Intermittent feeding allows remediation without well clogging.	Results of bench- and pilot-scale studies	Witt et al. (1999)
2. Develop an effective delivery system that spans entire width and depth of contamination.	Screen delivery wells over vertical extent of contamination (9.1–27.4 m bgs).	Long screens allow chemical and organism delivery at all depths.	CT measurements obtained from continuous cores during preliminary drilling	See text
3. Minimize costs.	Space delivery wells close to one another (1 m).	Short distances reduce time and volume for pumping. Pilot scale study demonstrated reliable cell transport over a 1 m distance.	Results of pilot-scale study, modeling results for tracer transport, cost optimization modeling	See text
	Circulate ground water at $Q = 9.1$ m^3/h for five hours each week.	25% breakthrough delivers adequate acetate for CT remediation.	Modeling results for tracer transport, confirmed with tracer studies; cost optimization modeling	See text
	Reverse circulation for one hour at the end of the weekly chemical delivery.	Reverse flow allows substrate delivery to regions around wells used during the five-hour extraction.	Modeling results for tracer transport	See text
	Alternate weekly five-hour extraction period between odd and even wells.	Alternate use of wells for injection and extraction will enable more uniform colonization.	Modeling results for tracer transport	See text

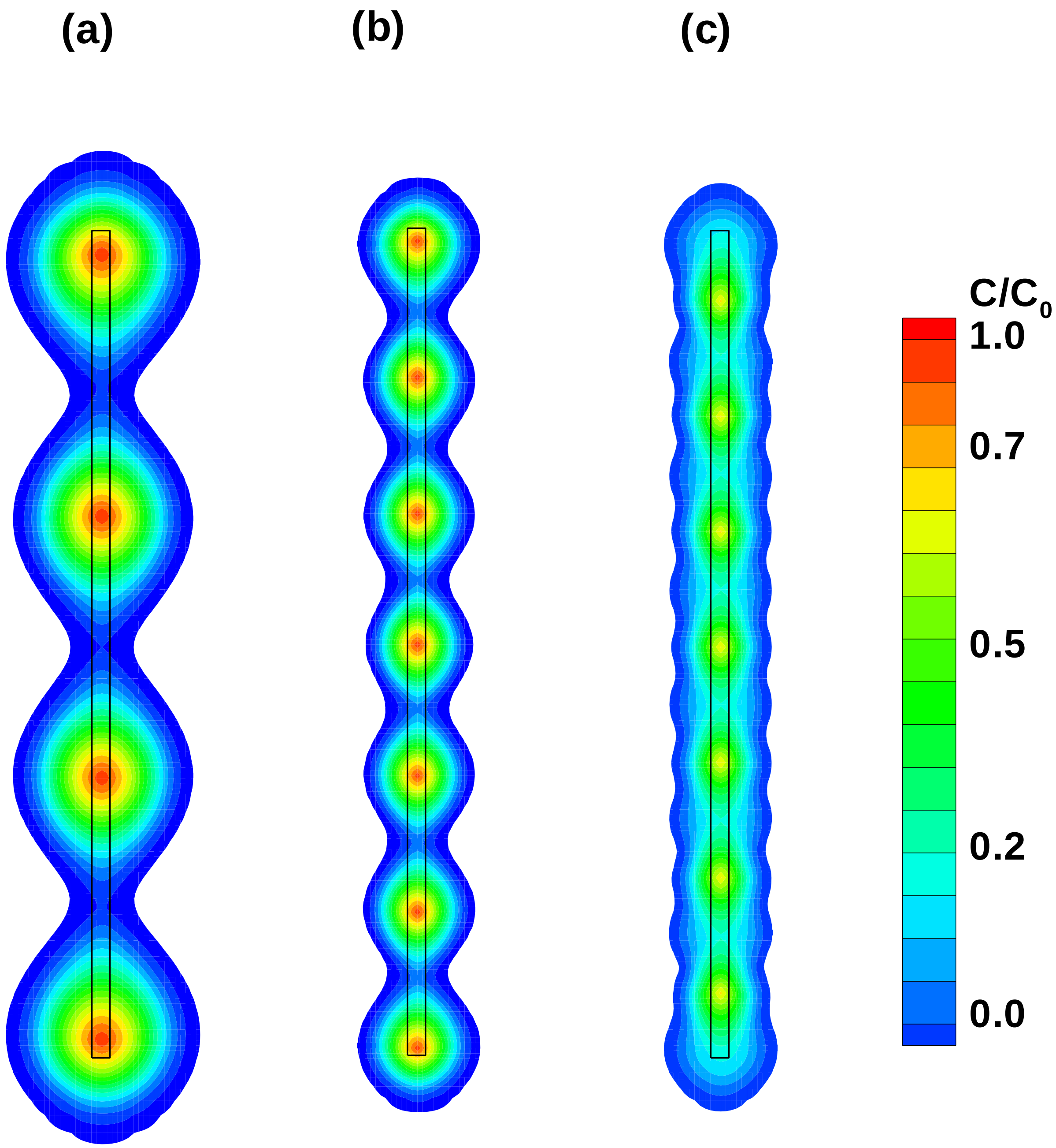

Figure 2. Simulated solute concentration fields for optimal delivery well design using (a) seven wells, (b) 13 wells, and (c) 15 wells with one hour flow reversal. All cases used 9.1 m^3/hr total pumping rate distributed across the delivery wells. The black rectangular domain represents the region where the concentration ratios were constrained to be above a specified concentration threshold.

imately 100 ppb), dissolved oxygen (2 to 5 mg/L), and nitrate (approximately 65 mg/L); is phosphorus- and carbon-limited; and has a pH of 7.2 to 7.4. Phosphorus levels and pH within a small test zone of the aquifer were increased to a range favorable for strain KC using pulses of base- and phosphate-amended ground water. Strain KC was grown in an aboveground reactor on-site and introduced into the test grid, along with a pulse of acetate and phosphate. In situ microbial activity was sustained with periodic pulses of extracted ground water amended with acetate, base, and phosphate. At those wells where base and acetate were effectively delivered, strain KC was detected in the ground water, nitrate levels decreased by 85%, and CT levels decreased by approximately 65% without a significant increase in CF. However, uniform delivery did not occur at all locations and at all times. One of the six monitoring wells exhibited notably different behavior with regard to delivery of tracer, organisms, nitrate, sulfate, and pH. Heterogeneous delivery of strain KC was also observed, with highest concentrations persisting within 1 m of the injection well. Evidently, heterogeneity and the overall lack of hydraulic control within the test grid resulted in suboptimal CT removal, compared with the much higher efficiencies achieved in bench-scale column experiments where hydraulic control was assured. Final borings indicated removal of 60% to 88% of the sorbed CT, little or no sorbed chloroform, and the presence of variable levels of strain KC cells on aquifer sediments. An important conclusion of this study was that scale-up efforts should focus on optimization of chemical and organism delivery.

Selection of a Multiple Delivery Well System

The primary goal of the work presented in this paper was to design and implement an effective biocurtain to remove CT that is transported with the ground water and sorbs to aquifer material as it passes through the biocurtain. To design an effective full-scale system, we had to overcome the chemical and organism delivery limitations of our pilot-scale experiment (previously described). To reduce the influence of heterogeneity, we initially considered construction of a "funnel and gate" system (Starr and Cherry 1994). In this approach, impermeable subsurface barriers are used to funnel ground water through a "gate," a region of enhanced chemical and organism delivery. This approach could potentially have reduced the size of the region over which chemical and organism delivery was necessary. However, solute transport simulations of different pumping scenarios for the gate design revealed that multiple closely spaced wells with long well screens would be able to deliver chemicals and organisms in a cost-effective manner, without the need for a barrier of uncertain performance. Accordingly, we selected the multiple well strategy for scale-up. In the following sections, we describe the design and hydraulic characterization of this system.

System Design

We began the design process at Schoolcraft with three major objectives: (1) achieve CT and nitrate effluent concentrations below the regulatory limits (5 ppb CT and 10 mg/L NO_3-N); (2) provide treatment over the entire vertical and horizontal extent of contamination in a plane perpendicular to the groundwater flow direction; and (3) minimize costs. The strategies we used to satisfy these objectives are summarized in Table 1, with an explanation for each strategy.

To satisfy the second two objectives, we had to overcome the constraints imposed by the depth of the CT plume and the heterogeneity of the aquifer. This was partially achieved by selecting delivery wells screened over the entire 15 m depth of contamination. Our goal was to intercept and treat the plume as it passed through a zone 15 m in depth and 15 m in length, and at least 0.3 m thick (in the direction of natural gradient ground water flow direction). To ensure delivery of the necessary chemicals and organisms across this zone, we applied optimization methods coupled with flow and transport models.

At the time of system design, we had incomplete information on aquifer heterogeneity and potential costs associated with various decisions. We were obliged to select the number of wells and initial flow rate using scientific judgment and optimization simulations with incomplete data. Subsequently, we obtained improved data (costs for manpower and the larger pipes and pumps at moderate flow rates), and we refined and formalized the objective function (1) used for optimization simulations. The optimization work described next indicates that the number of wells that we selected was optimal. It also indicates that our choice for the pumping rate was suboptimal but reasonable given other constraints.

Other constraints on the chosen system design should be mentioned. To achieve maximum symmetry of the biocurtain feeding zone about the center well, we constrained the system to an odd number of wells. We also set the pumping frequency at once weekly based on laboratory (Witt et al. 1999) and pilot-scale results (Dybas et al. 1998). The resulting mixed integer programming problem (Floudas 1995) was solved as a series of optimizations, each with a fixed number of wells across the treatment zone width. To demonstrate that treatment could be achieved with less pumping than a pump-and-treat system, we rejected optimal solutions if the total pumped volume during each pumping period exceeded the natural gradient ground water flow through the biocurtain over a one-week period.

The goal of optimization modeling was to find the most cost-effective method of introducing substrate into the subsurface so that a specified percentage p (i.e., 100%) of the cells in the biocurtain region would have concentration ratios (C/C_0) above a specified threshold value C_{th}, where C_0 is the injected solute concentration. This goal was selected so that microbes located between adjacent delivery wells would receive enough substrate (acetate) during weekly pumping events to degrade all of the CT and nitrate entering the treatment zone during the following week. In addition to the total flow rate Q, which is the independent variable, the parameters that affect the objective function include the threshold concentration value C_{th}, the number of wells N_w, the number of pumps N_p, the pumping cost P, and the substrate cost D. Both the pumping cost and the substrate cost are functions of flow rate and event pumping time (t). This constrained optimization problem can be posed as follows.

Minimize

$$f(Q) = T \cdot [P(Q,t) + D(Q,t) + m(t)] + A(N_W) + B(N_p) \quad (1)$$

subject to

$$Q_{min} \le Q \le Q_{max}$$

$$C_i(Q,t) \ge C_{th} \; \forall\, i \in \Gamma, \; \sum_i \frac{i}{i_{max}} \ge p$$

$$\{N_W, N_p\} \in I$$

$$N_W \in O \qquad Nw \le 15$$

where

T	=	number of pumping events in the remediation process (= 1300 for a 25 year period with one pumping event per week)
P	=	pumping cost = $C_R \cdot Q \cdot t \cdot E$
C_R	=	a fixed ratio used to determine the power usage of pumping (1.5);
E	=	unit energy cost (≈\$0.06/kw-hr);
D	=	cost of injected substrate = $M(Q,t) \cdot F$
M	=	mass of injected substrate for one pumping event
F	=	unit cost of substrate (≈\$0.82/lb)
m	=	manpower cost (≈\$100/hr)
NWMAX	=	maximum number of wells = 15
NPMAX	=	maximum number of pumps = 8
A	=	one-time construction and installation cost for the wells' piping systems, which can be described using the following equation that includes a N_W dependent term based on quotes from our drilling contrac-

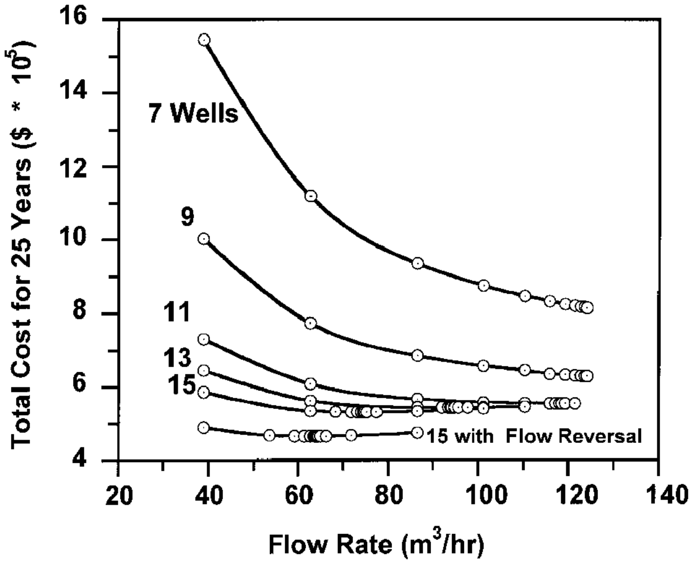

Figure 3. Plots of expected system installation and operation costs as a function of the number of wells and pumping rate. The total cost for 25 years of operation include the long-term manpower cost and the system installation cost, which was used to determine that 15 wells is a good long-term strategy.

tor and a Q dependent term due to higher costs for pipes, valves, and pumps, for higher flow rates based on a fit to consultant quotes: [≈$11,300 +$90,000 $(N_W/60)^{0.6}(Q/9.1)^{0.7}$]

B = cost for pumps ($2500 per pump) which are shared between injection and extraction wells, thus $N_p = (N_W/2) + 1$

$C_i(Q,t)$ = concentration of tracer at the ith cell in a predefined region, Γ

Q_{min}, Q_{max} = lower and upper bounds for the pumping rate (0.4 m^3/hr to 125 m^3/hr)

i_{max} = total number of cells in the domain

I = set of integers

O = set of odd integers

p = specified percentage of cells to satisfy the threshold condition in the biocurtain region, Γ

The nonlinear objective function (1) was minimized using a one-dimensional golden section search algorithm (Press et al. 1995). This procedure adjusts Q and N_W to achieve a minimum cost scenario. We selected this algorithm because it can rapidly find the global minimum even in cases where local minima exist, without relying on the derivative of the objective function.

The objective function was evaluated using flow rates and concentrations obtained from two-dimensional tracer concentration simulations (discussed in the "Tracer Test" section). The governing equations for ground water flow and tracer transport were solved for each selected flow rate. Each simulation continued until the concentration ratios (C/C_0) of 100% of cells in the biocurtain region (Γ) exceeded the specified concentration threshold (C_{th}) of 0.25. For the chosen geometric constraints on the treatment zone, the corresponding flow rate and time were used to calculate the cost for the specified pumping rate. Figure 2 shows the computed solute concentration plumes for seven, 13, and 15 wells, with a box around the predefined biocurtain region where concentration values are checked.

As shown in Figure 3, the 15 well scenario had the lowest cost for all pumping rates, and the optimal flow rate is 63 m^3/hr for approximately 0.7 hours of pumping and 0.1 hours of flow reversal. The 15 well scenario also provided a well spacing of roughly 1 m, which is a distance over which high levels of microbial colonization of sediment had been observed at the pilot-scale site (Dybas et al. 1998). Without adequate microbial colonization of the space between the delivery wells, gaps would be present in the biocurtain. Although Figure 3 suggests that further cost reductions would occur with more than 15 wells, we did not explore this possibility because of the practical limitations in drilling wells with a spacing less than 1 m to 27 m depth. Based on this initial optimization work, we installed a row of 15 wells (Figures 4 and 5) coupled to an above-ground chemical delivery and mixing system.

To estimate desired flow rates for system operation, we considered several pumping scenarios. These included injection into all wells with no extraction, and alternating injection/extraction wells with a second phase in which the flow direction is reversed (injection wells became extraction wells and vice versa). The inject-only scenario was rejected because the addition of water from outside the delivery gallery would make interpretation of remediation efficiency more difficult. During the two-phase simulation, we switched to a flow reversal mode when 80% of the cells in the biocurtain region exceeded the concentration threshold of 0.25, and again ended the simulation when 100% of the cells met this constraint. Flow reversal ensured substrate delivery to regions around the five-hour extraction wells.

To achieve the threshold concentration across the delivery well gallery, the number of delivery wells can be increased or the average volume pumped can be increased. Figure 6 illustrates this trade-off, where the average volume pumped is calculated using the flow rates from Figure 3 multiplied by the pumping times needed to reach the threshold concentration. As expected, the volume required to ensure adequate substrate delivery is primarily a function of the number of wells, and it is a weak function of flow rate (less than a 3% volume variation across the imposed flow rates). For 15 wells, the average volume pumped to achieve the threshold concentration is 51.1 m^3, close to the 54.5 m^3 chosen for system operation. At the time of system installation, we had evaluated the feasibility of pumping rates in the 40 to 125 m^3/hr range and found that the engineering costs for piping, valves, and pumps were prohibitive. Accordingly, we limited our flow rate to 9.1 m^3/hr. The natural gradient flow through the biocurtain was estimated at approximately 80 m^3/week using our ground water flow model. At the chosen pumping rate, approximately 68% of the natural gradient flow is pumped to the surface for chemical addition. To pump 54.5 m^3, we pumped 45.4 m^3 in the first five hours, reversed flow direction, and pumped 9.1 m^3 in the final hour. This ratio was confirmed by subsequent optimization work. The rationale for using a five hour initial pumping period is further discussed in the "Tracer Test Data and Modeling" section.

According to our most recent cost analysis of the 15 well scenario (Figure 3), we are operating our system at a slightly suboptimal pumping rate. However, once our system was installed, the potential marginal cost reductions for increased flow were not practical. In any case, the optimization work confirmed that our system is cost effective relative to other pumping scenarios. Our analysis also indicated that a row of 15 closely spaced wells provide fairly uniform chemical delivery across a plume transect using short pumping periods and low flow rates. By extending the length of this 15 m well gallery, we could cover the entire plume width. Also, by increasing flow rates during weekly pumping events, we

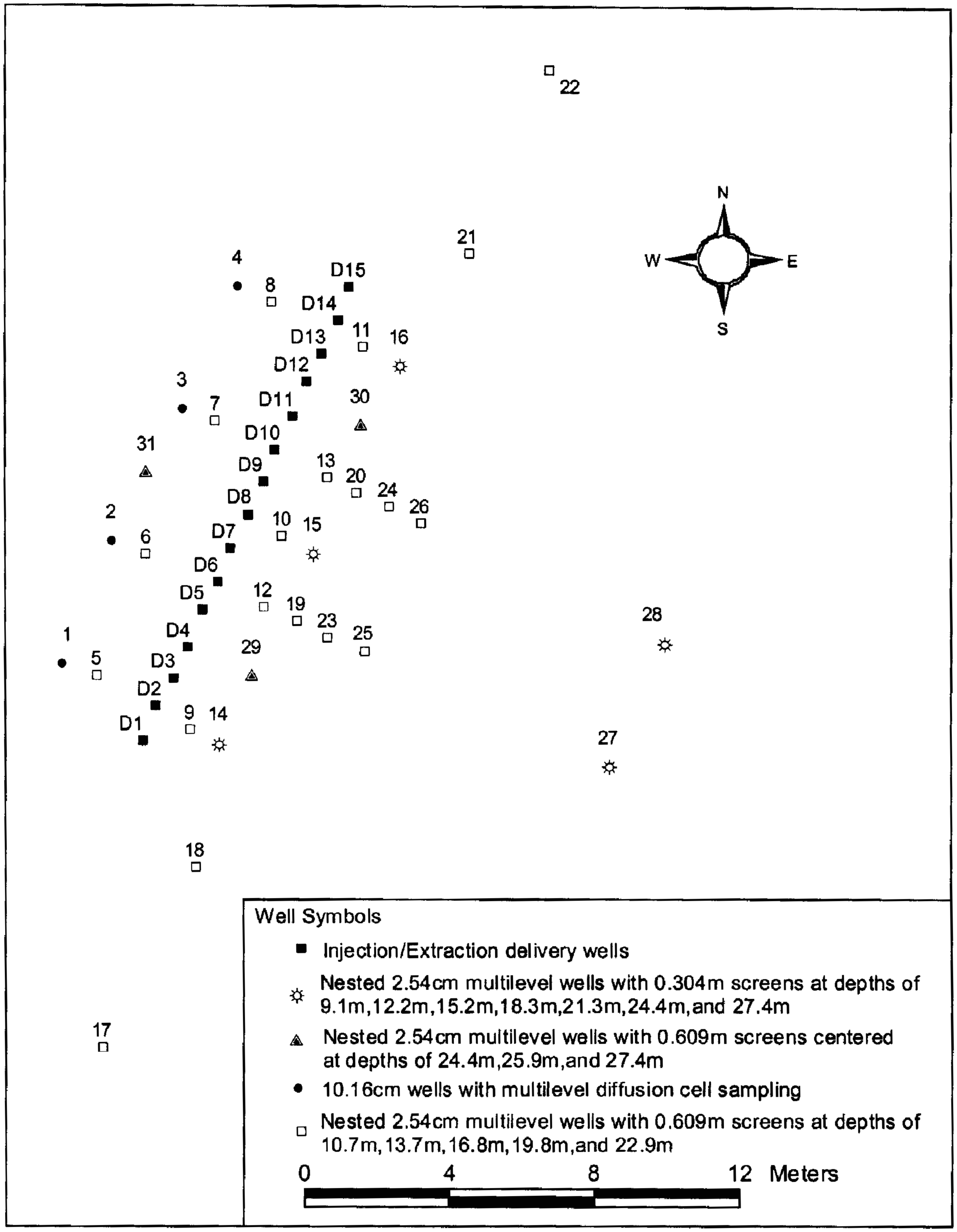

Figure 4. Map of the installed delivery wells (D1 through D15) and monitoring wells (1 through 28) at the Schoolcraft Plume A site.

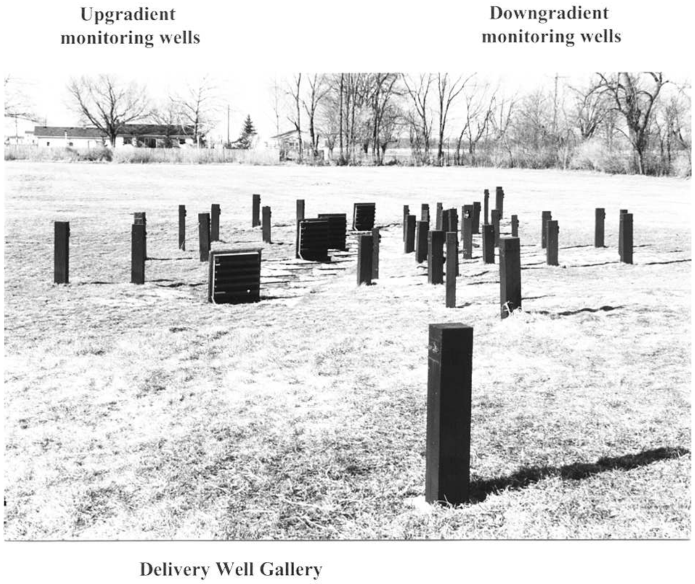

Figure 5. Photo of the site showing the delivery well gallery in buried enclosures, and upgradient and downgradient monitoring wells.

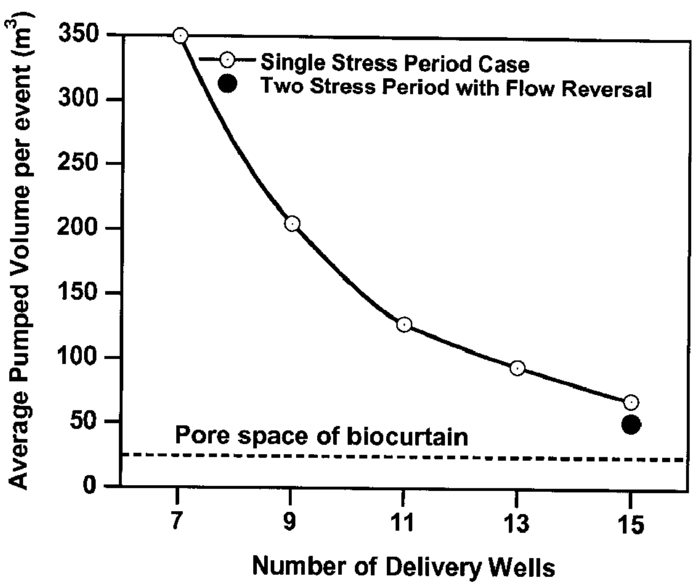

Figure 6. Average of the optimal total flux in a pumping event for the range of pumping rates shown in Figure 3. The total flux is relatively constant across the range of pumping rates for each well scenario.

could shorten the time required for pumping events, decreasing labor and total costs.

Characterization of the Treatment Zone and Installation of the Delivery Wells

During construction of the delivery well gallery in the fall of 1997, we collected sediment cores for visual classification and aquifer property measurements. We collected seven continuous cores between 9.1 and 24.4 m depth during drilling of the even numbered delivery wells (Figure 7) using a Waterloo cohesionless continuous sand sampler as described in Dybas et al. (1998). To obtain minimally disturbed samples in a 5 cm diameter plastic sample liner, a 14.6 cm O.D. auger was advanced in 1.5 m segments, and the 1.5 m long sampler was hammered into the aquifer ahead of the auger. A piston in the sampler provided vacuum to help retain the sample during recovery. After removing subsamples for volatile analysis, the core segments were transported vertically on ice to the

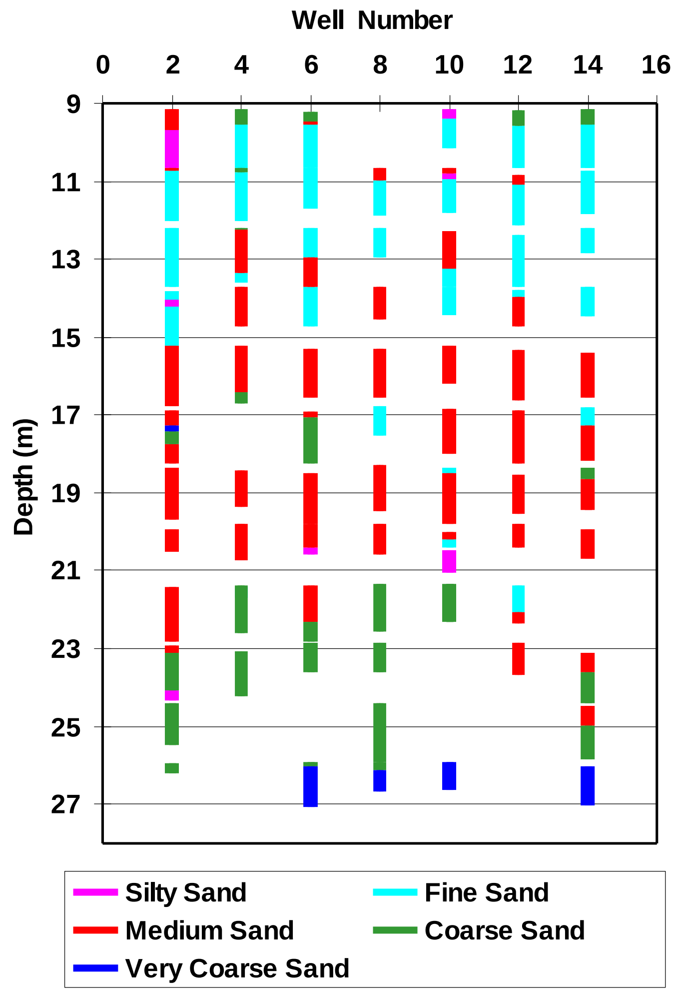

Figure 7. Visual classification of the collected core into five groups.

environmental engineering laboratories at Michigan State University for storage at 10°C. For the sampled aquifer cross section, we collected 122 m of core liner with 85 m of recovered sediment core (Figure 7). Subsamples from each of 220 core segments were analyzed for hydraulic conductivity, sorbed contaminant concentrations, and sorption parameters. This permits characterization of mean property values as well as property variability.

To obtain high-resolution hydraulic conductivity measurements of the biocurtain treatment zone, we first visually inspected the cores in the laboratory and classified them into five groups (silty sand, fine sand, medium sand, coarse sand, and very coarse sand) as shown in Figure 7. We then cut the cores into 20 to 25 cm segments, removed the sediment from the liners, dried them at 60°C for 48 hours, and stored them in labeled plastic bags at 20°C until analysis. For permeameter measurements, we removed large stones (diameter greater than 20 mm), homogenized the sediment, packed 300 grams of dried sediment into a permeameter, and saturated the sample from the bottom after flushing with CO_2. For each sample, hydraulic conductivity was measured by passing Schoolcraft ground water, which was deaired under a 400 mbar vacuum for 30 minutes, through a constant head permeameter. The aquifer materials from 9 to 27 m bgs have high hydraulic conductivity values with an average of approximately 0.027 cm/s and a log(K) variance of 0.12, with the highest conductivity sediments near the bottom of the aquifer. As shown in Table 2, the highest conductivity sediments were

Table 2
Measured Hydraulic Conductivity Values Averaged in 3 m Thick Layers

Depth (m)	9–12	12–15	15–18	18–21	21–24	24–27
No. of Measurements	40	35	42	36	36	31
Average K (cm/sec)	0.0107	0.0115	0.0240	0.0182	0.0389	0.0525
Average log K (cm/sec)	–1.97	–1.94	–1.62	–1.74	–1.41	–1.28
Variance log K	0.0349	0.0160	0.0413	0.1824	0.0530	0.0332

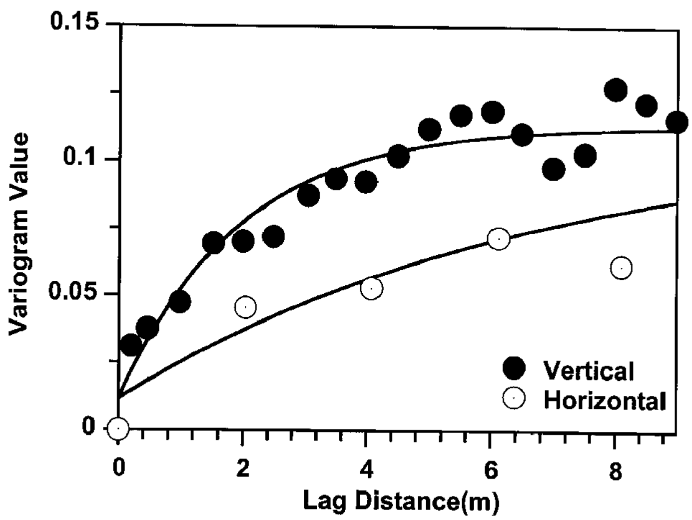

Figure 8. Experimental and model variograms of the log(K) data illustrating a significant anisotropy in correlation length.

Table 3
Layered Hydraulic Conductivity Values Used to Predict Tracer Concentrations from Extraction Wells in the Delivery Well Gallery

Layers	Top Elevation	Bottom Elevation	K (cm/s)
1–13	2.5	15.5	0.011
14–19	15.5	21.5	0.027
20–22	21.5	24.5	0.055
23–25	24.5	27.5	0.040

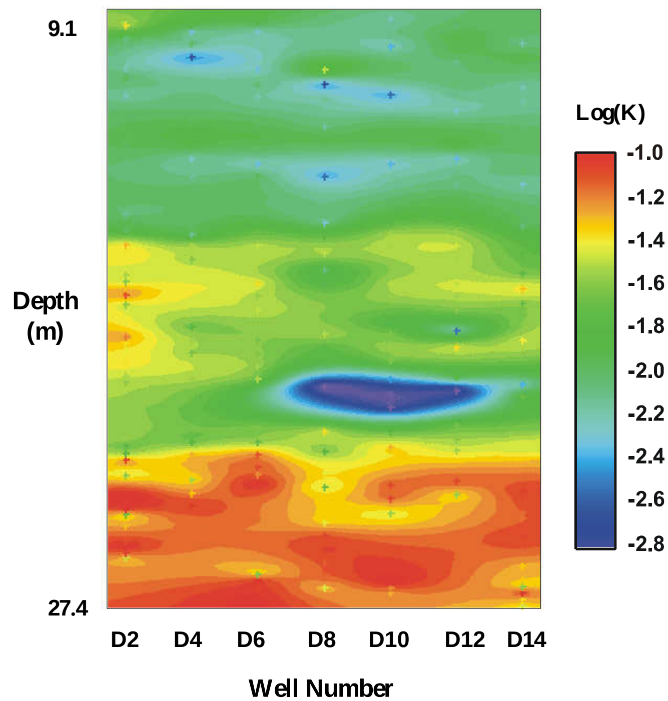

Figure 9. Two-dimensional map of log (hydraulic conductivity) values developed using ordinary kriging and the variograms from Figure 8 to interpolate between 220 measured hydraulic conductivity values described in Table 2.

located from 21 to 27 m depth, near the bottom of the aquifer where coarse sands were observed.

Based on the optimization work discussed previously, we installed a delivery well "gallery" consisting of 15 wells in a row that transects a 14 m lateral section of the downgradient edge of Schoolcraft Plume A (labeled as wells D1 through D15 in Figure 4). The delivery wells were installed approximately 1 m apart using 33 cm diameter hollow-stem augers, and were screened from 9.1 to 24.4 m bgs (using 0.025 cm slotted screen) to span the anticipated thickness of the contaminant plume. A pair of adjacent 5 cm diameter wells was installed at each delivery well location. One of these wells served as the delivery well, and the second well served as a monitoring well during pumping events and as a back-up delivery well in the event of well clogging or breakage. The well-screen pair in each borehole was joined by 2.5 cm spacers positioned every 1.5 m throughout the screened interval. After installation, the auger was pulled allowing natural sediment collapse around each screen and casing.

Each delivery well was connected by flexible tubing through a valve manifold to eight suction pumps that discharge to an above-ground mixing and injection system. The pumps were primed using a 1.5 hp vacuum pump, with any extracted granular material removed using basket strainers upstream of the pumps. The above-ground system includes a high flow rate recirculation loop with mixing devices for substrate and other chemical delivery, and two 9.46 m^3 fermentation vessels for aboveground microbial inoculum growth. Although nutrients are added to the water before injection, contaminants are not removed from the system by any treatment before reinjection. Thus, the injected water has contaminant concentrations that roughly equal the average extracted concentration. This system and the biodegradation reactions will be described in more detail in a later paper (Dybas et al., submitted) that addresses inoculation and the results of long-term system operation.

Installation and Sampling of the Monitoring Well Grid

A detailed sampling grid was installed to monitor changes in solute concentrations in the remediation test site. This grid included multiple depth-specific monitoring points (96 downgradient, 23 upgradient, and 20 outside the treatment zone). The upgradient points allowed us to monitor changing influent solute concentrations, while monitoring wells 17, 18, 21, and 22 provided downgradient background concentrations and allowed for interpretation of the lateral extent of the treatment zone.

Two types of monitoring wells were installed. The majority of the wells in the grid are nested clusters of 2.5 cm diameter wells with 0.33 m long screens (0.025 cm pre-cut slots) at the depths shown in the legend of Figure 4. Four 9 cm diameter monitoring wells (1, 2, 3, and 4) were installed to accommodate multilevel diffusion cell sampling (MLS) devices (Margan Ltd. Israel) by drilling to 24.4 m bgs with 25 cm diameter hollow-stem augers (20.5 cm I.D.). Well screens with 0.025 cm pre-cut slots were positioned between 9.15 and 24.4 m bgs. Native formation materials were allowed to collapse around all monitoring well screens and casings, except for roughly the top 3.5 m bgs, which were backfilled with a mixture of granular bentonite and sediment/soil cuttings.

Each monitoring well was developed by pumping 40 L at 2 L/min, and finished by installing 1.6 cm diameter high density polyethylene (HDPE) drop pipes that were sealed 15 cm above each screened interval. Drop pipes were installed to reduce the purge volume that had to be removed to clear the casing during ground water sampling. The drop pipe was fitted above ground with a flexible length of TygonT fuel and lubricant hose chosen for its low organic sorption and flexibility over a broad temperature range. The monitoring wells were sampled with a series of Masterflex multihead peristaltic pumps after purging for 30 minutes at 200 mL/min. In a typical monitoring event, samples from 50 to 139 discrete depth intervals were obtained.

Geostatistics

In order to incorporate our hydraulic conductivity dataset into our ground water flow and solute transport models, we surveyed the surface location of each well using a LeicaY Total Station and corrected the sample location depths to a common reference. We then developed experimental variograms (Figure 8) to estimate the vertical and horizontal correlation lengths from the log(K) values measured on the core samples. The log(K) data were used in this and subsequent geostatistical analyses because hydraulic conductivity is commonly considered a log normally distributed parameter (Hoeksema and Kitanidis 1985). The parameters for the horizontal and vertical correlation structures were estimated to fit the experimental variograms. The variograms in Figure 8 show a strong anisotropy for the Schoolcraft site, with a horizontal correlation length of 20.8 m, vertical correlation length of 5.8 m, variance of 0.11, and a nugget of 0.011.

Using the estimated correlation parameters, we generated a two-dimensional map of hydraulic conductivity across the delivery well gallery (Figure 9) using ordinary kriging to interpolate between closely spaced samples (Sudicky 1986; Deutsch and Journel 1998). As illustrated, the hydraulic conductivity of the Schoolcraft sediments varies by more than an order of magnitude with significant spatial structure across the measured vertical section through the aquifer. The variability in hydraulic conductivity in Figure 9 is geologically reasonable, with significant variability around a general layered trend. The variations are especially notable near the low conductivity deposit at approximately 21 m depth in both the visual classification (Figure 7), and in the interpolated two-dimensional map of measured log hydraulic conductivity (Figure 9).

Tracer Test Data and Modeling

Predictive Tracer Transport Modeling

To evaluate the influence of heterogeneity, we developed and updated flow models as data on the heterogeneous aquifer properties became available. A three-dimensional solute transport model with layered hydraulic conductivity values was developed to predict the tracer concentration history for a representative extraction well, which we used to evaluate the duration of pumping at 9.1 m^3/hr that would be required during system operation. We used a layered model because only a subset of the hydraulic conductivity mea-

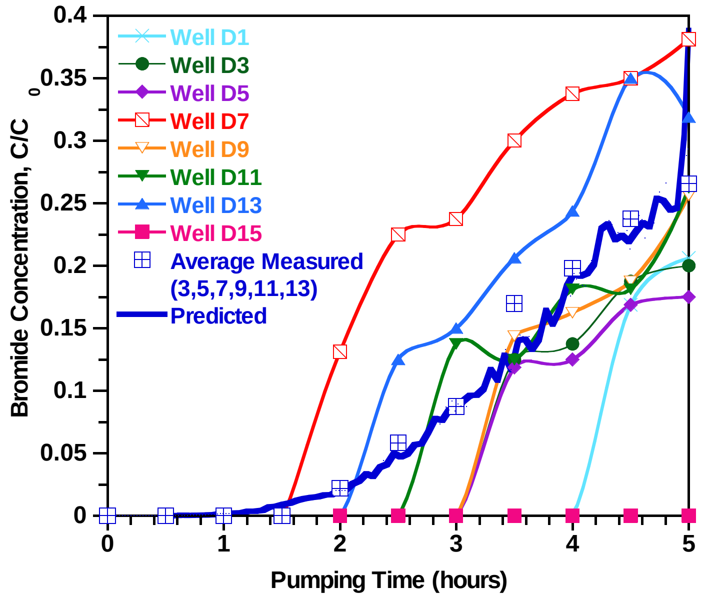

Figure 10. Observed Br concentration histories from the extraction wells (D1 through D15) and an average of observed Br concentrations from extraction wells D3 through D13, which can be compared to the predicted concentrations based on a layered hydraulic conductivity model. The injected Br concentration was 16 ppm.

surements described (Table 2) was available at the time we needed to decide on system operation. The geometry of the conductivity layers was determined by visually separating a map of conductivity values into four layers and assigning mean values to each layer (Table 3). To better represent the vertical variability in solute concentrations, we assigned the values listed in Table 3 to 25 1 m thick layers. A representative longitudinal dispersivity of 10 cm and a ratio of transverse to longitudinal dispersivity of 0.2 were used based on Neuman (1990).

Our transport model simulates steady-state ground water flow and nonreactive scalar transport by numerically integrating Darcy's law coupled with the unsteady advection-dispersion equation in three dimensions. These equations were solved subject to constant head boundaries on the upgradient and downgradient ends of the model that provided the correct water table elevation of approximately 4.5 m bgs at the delivery well grid while inducing a gradient of 0.0011 based on regional head measurements (Figure 1). No-flow boundaries representing flowpaths were used on the sides of the model.

The model domain is a rectangular region of 102 m 2 57 m 2 23 m extent, which we discretized into a computational grid composed of 134 cells in perpendicular to natural gradient direction, 86 cells in the natural gradient flow direction, and 25 m thick layers in the vertical dimension. The delivery well gallery is located at the center of the computational domain, where fine cells (20 cm 2 20 cm) that are roughly equivalent to the diameter of the delivery well boreholes were used. The cell size was increased in a geometric progression away from the gallery toward the boundaries. We used the three-dimensional finite-difference ground water flow model MODFLOW, of the U.S. Geological Survey (McDonald and Harbaugh 1988), to compute hydraulic heads for the region. We then solved the advection-dispersion equation using MT3D with fourth- order Runge-Kutta particle tracking and the hybrid method of characteristics (Zheng 1992).

We used this transport model to predict the tracer concentrations and evaluate the pumping duration specified by the single layer optimal delivery design model, without any calibration to field data. Our goal was to predict the pumping time necessary to achieve a desired level of tracer breakthrough ($C_{extract}/C_{inject} = 0.25$). This extraction to injection concentration ratio was calculated as a conductivity-weighted average across the screened interval of the delivery wells. This criterion was chosen because it was an easily measured quantity that is consistent with our objective function. For the specified pumping rate of 9.1 m^3/hr, the model predicted that the $C_{extract}/C_{inject} = 0.25$ would be achieved in the central extraction wells after approximately five hours of pumping as shown by the solid line in Figure 10. The optimization study described previously indicated that we could achieve a more even distribution of tracer and further reduce operating costs using a second one-hour phase when the pumps were reversed. Flow reversal allowed us to achieve more uniform solute concentrations between the delivery wells, by raising concentrations near the former extraction wells where they would otherwise have been too low. This five hour flow/one hour flow reversal scheme was selected as our operational field pumping strategy.

Field Tracer Test

In November 1997, we performed a full-scale tracer test to verify that our selected pumping strategy would provide roughly the expected extraction well concentration histories and to evaluate the fate of fluid pumped through the delivery well gallery. Tracking the tracer concentration front through the well gallery provides a measurement of the natural-gradient solute travel times through the sampling grid. Measured changes in contaminant concentration concurrent with tracer arrival can be attributed to factors such as the mixing of contaminants across the screened interval. Flourescein and bromide were dissolved in the injected fluid during the first injection-extraction pumping event, followed by a period of 19 days with no pumping to allow natural gradient tracer transport. This test provided critical information about the effectiveness of the pumping system, heterogeneity downgradient of the injection wells, the expected distribution of solutes across the delivery well gallery, and the influence of pumping on contaminant concentrations prior to bacteria injection.

The initial tracer test was conducted using the preselected pumping scenario, with a five-hour period of 9.1 m^3/hr injection containing 130 ppb flourescein dye and 16 ppm bromide among the even numbered injection wells, with simultaneous extraction from the odd numbered delivery wells. Tracer was injected for the one-hour flow-reversal phase using the same pumping rate but extracting from the even numbered wells and injected into the odd numbered wells with average injected concentrations of 22 ppm for Br and 215 ppb for fluorescein. The ratios of the injection from the second phase to the first changes because the bromide is contained in a tank with sodium hydroxide, which was used for pH adjustment, and its addition was controlled by a pH feedback loop. In contrast, the flourescein was injected at a constant rate into the injection line with no feedback. Since the flow reversal phase gets its extracted water from wells that were injecting fluid for five hours, this flow contributes significant tracer mass to the injection lines. The fluorescein concentration nearly doubles, due to the constant injection plus extraction of ground-water with 130 ppb flourescein, while the bromide stayed relatively constant due to the pH control. Flourescein and bromide were chosen because our laboratory experiments showed that these tracers are relatively conservative, do not affect bacterial activity or contaminant degradation, and are easy to detect. Flourescein was injected as single pumping event slug to observe the behavior of a pulse of water as it moved through the monitoring grid, while bromide was injected with every slug of injected water to tag all injected water with an independent tracer.

The predicted tracer concentrations were nearly equal for the extraction wells (except that wells D1 and D15 extract less tracer because they are located on the ends of the well gallery) because this simulation used the layered conductivity model with equal well spacing. Figure 10 illustrates the simulated concentration history at well D7 and the measured concentration histories at each extraction well. The concentrations in Figure 10 are plotted with respect to time elapsed since the initial tracer injection. At the time of this predictive simulation, only enough conductivity data were available to develop a model with homogeneous conductivity layers, thus we compare the simulated concentrations to the average of the observed concentration histories from the central extraction wells (D3, D5, D7, D9, D11, and D13). The model accurately predicts the averaged observed concentrations with no parameter adjustment, as shown in Figure 10. This modeling-based prediction was used to design the pumping scenario before the tracer test was conducted, and no parameter values were adjusted to fit the simulation to any field measurements.

The success of this model prediction illustrates the value of hydraulic conductivity data for solute transport simulations. The differences between the individual measured concentration histories at the extraction wells indicate that either the heterogeneity between wells or the variable offset between injection and withdrawal wells have significant effects on the tracer transport during the injection period. The homogeneous layered model adequately predicted the average measured tracer transport across the delivery well gallery during a pumping event.

Measured Carbon Tetrachloride Degradation

A complete sampling of the observation wells in Figure 1 was performed in June 1999 to evaluate system performance. Remediation percentages were calculated relative to 31.8 ppb, which is the average CT concentration measured at the extraction wells after one hour of pumping during two events: one prior to inoculation (31.9 ppb on March 18, 1997), and one two weeks after inoculation (31.7 ppb on January 12, 1998). The estimated remediation percentages for the 63 observation points downgradient of the injected biocurtain (between depths 10.7 m and 22.9 m, laterally from well 9 to well 11 and downgradient to wells 24 and 25) are illustrated in Figure 11. The remediation percentages for this region have a mean of 95.6% and a median of 98.5%, with most locations showing between 95% and 100% efficiency. During the June 1999 sampling event, CT concentrations downgradient from the same sample points ranged from 0.025 to 10.88 µg/L, with a median of 0.46 µg/L. During this event, 58 out of these 63 observations were below the 5 µg/L regulatory standard for CT.

The influent concentrations increased over this multiyear period since our system is near the downgradient edge of the CT plume. For example, the average CT concentrations from wells 5, 6, 7, and 8 at the 22.9 m depth, from which a significant proportion of our extracted water is derived due to high conductivities in this region, increased from 32.5 ppb on February 6, 1998, to 52.3 on April 27, 1999. A full discussion of the biocurtain system performance will be provided in a future paper (Dybas et al., submitted).

Discussion and Conclusions

The design that we implemented at the Schoolcraft site is one of the first examples of an effective long-term full-scale biocurtain. The design and some aspects of the design process appear suitable for application to other sites. A key step in the design process was selection of an optimal well spacing based on regional knowledge of the aquifer and pilot-scale data. This allowed us to explore the potential efficiency of a wide range of well spacing and pumping alternatives before local hydraulic conductivity data were available. The wellfield geometry and pumping rates were designed to minimize the cost of installation and 25 years of operation. Simulations indicated that closely spaced wells with intermittent pumping would permit effective delivery of the agents needed for remediation across the aquifer despite significant variability in hydraulic conductivity. These predictions were confirmed experimentally.

As discussed earlier, the initial design optimization study was completed before the data indicating significant heterogeneity in hydraulic conductivity were available. This is a common situation because detailed data is typically not available until after the wellfield installation. In our case, we used a representative hydraulic conductivity from the pilot-scale experiment (0.032 cm/s) to develop a one-layer transport model. By coupling this model to our optimization algorithm, we established basic design parameters (well spacing, number of wells). We found that closely spaced wells enabled lowest long-term operational costs, and ensured effective distribution of the injected solutes and microbes. Subsequently, we discovered that hydraulic conductivity varied by over an order of magnitude, and we realized that this variation dramatically influenced flow and transport. It is interesting to note that, in this case, prior knowledge of heterogeneity would not have had a significant impact on system design. The long-term cost function (Figure 3), shows that cost is a smooth function of the pumping rate and a strong function of the number of wells. If a heterogeneous flow field is simulated, we expect the closely spaced well scenario to have the lowest cost solution because the heterogeneous conductivity field can be thought of as a generalized perturbation around the mean value used in our optimization study. In fact, the cost advantage of closely spaced wells is likely more significant in a heterogeneous aquifer because the volume of water needed per pump event will be higher to deliver nutrients across the entire vertical extent of the biocurtain.

Despite significant heterogeneity in hydraulic conductivity, we demonstrated that relatively simple ground water flow and solute transport models can provide accurate predictions of the average conservative tracer transport during pumping events. The purpose of the three-dimensional flow and transport model with homogeneous conductivity layers was to predict the transport of tracer between injection and extraction wells based on a limited set of hydraulic conductivity data. This simulation allowed us to develop a pilot operating scheme by establishing a minimum threshold for breakthrough at extraction wells (C/C_o of 0.25). This ensured that most of the biocurtain region would be exposed to the injected solute. Modifying pumping rates could satisfy this criterion, although that was not necessary in this instance. Based on the predictive simulation, we chose to use five hours of paired extraction and injection followed by one hour of reversed flow. The model provided an excellent prediction for average breakthrough behavior with no adjustment of simulation parameters. As a result, we continued to operate under this pumping regime. Thus, despite the variable hydraulic conductivity, a layered transport model prediction allowed for effective design of our pumping scheme.

Our design evolved from a funnel and gate concept to the installed set of closely spaced wells equipped with screens spanning the vertical extent of contamination. A distinctive aspect of this design was the use of short pumping periods (only six hours per week) through long well screens. This weekly operational cycling was derived from bench- and pilot-scale studies. Thus, our optimization strategy implicitly incorporated constraints defined by an interdisciplinary team of scientists and engineers. The design progressed through many possible designs as new data became available, or as model simulations provided new insight. We would not have been able to develop our final system without detailed aquifer characterization efforts, high-resolution numerical modeling, or the interdisciplinary design meetings that shaped the project.

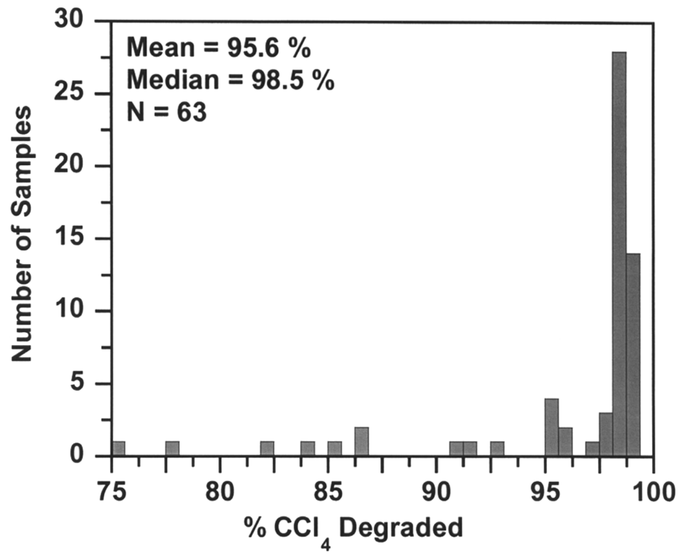

Figure 11. Carbon tetrachloride remediation percentages during mid-June 1999 at the 47 downgradient observation points that were also sampled in the week prior to inoculation (January 7, 1998).

A cost estimate for a full-scale system of this type spanning the entire width of Schoolcraft Plume A is approximately US$1.3 million, with the 25-year operation and maintenance accounting for approximately 60% of the costs (EFX Systems Inc., personal communication). For comparison, the treatment cost using a conventional approach (pumping to extract the water followed by air stripping) was estimated at approximately $6 million for a 25-year cleanup plan. We anticipate that automation may be used in future systems to further reduce operating costs.

The design and some aspects of the design process appear to be generalizable for application to other sites. We have demonstrated that the design is effective despite over an order of magnitude of variability in hydraulic conductivity. Since fully screened wells extract and inject water in proportion to the hydraulic conductivity along the screened zone, the design should effectively create a biocurtain that is thick in the high conductivity zones and thin in the low conductivity zones. This should create the desired effect of similar residence times for natural gradient flow through the biocurtain. For the presented case, the developed technology effectively delivered bacteria and nutrients across a fairly deep contaminated zone where it can be difficult and expensive to use alternative technologies such as emplaced reactive curtains. We anticipate that the system will operate efficiently for long periods of time, given that it has operated for approximately two years without a noticeable change in efficiency despite seasonal changes in water table elevation and increasing CT concentrations flowing into the biocurtain.

This is the first in a series of papers that will describe aspects of the field-scale bioaugmentation experiment at Schoolcraft, Michigan. Other papers will examine the long-term results of biocurtain operation (95% to 100% CT removal for more than two years), methods of estimating transport properties using geophysical and hydrogeological data, and the role of heterogeneity on flow and reactive transport in the region downgradient of the biocurtain.

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