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 In the last few years, researchers have developed several technologies to remediate MTBE. These have included advanced oxidation processes (AOPs) that use Fenton's reagent, ultraviolet light, UV/hydrogen peroxide, and ozone/hydrogen peroxide. Conventional techniques such as activated carbon adsorption and air stripping have not always worked as well as AOPs. AOPs, as discussed in this article, have successfully degraded MTBE.
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Experiences with pump-and-treat

This article summarizes some pump-and-treat systems installed at Superfund sites. Analysis of these systems will show that

- inadequate or nonconservative groundwater extraction rates have resulted in increased costs from additional investigation, design, construction and increased government oversight
- downtimes and maintenance problems have resulted in significant reductions in average groundwater extraction rates
- actual groundwater extraction rates are routine parameters that must be monitored and reported

Designing a P&T (pump-and-treat) system

The design focus of a P&T system is to create an acceptable capture zone. "The required capture zone for groundwater cleanup is generally defined as that part of the aquifer where groundwater contaminant concentration exceeds cleanup requirements, minus any area that will be addressed by monitored natural attenuation" (Boice, 2002). Remediators will generally also conduct water level surveys to define groundwater gradients and aquifer tests to assess overall hydraulic conductivity, which is probably one of the most uncertain input parameters of a groundwater model. *Reviewer's comment: Another focus of a P&T system would be capture of the plume's leading edge.*

Slug tests can be used to measure hydraulic conductivity. However, they are less accurate and representative than pump tests.

Remediators then model the system, using a steady-state groundwater flow model to determine minimum groundwater extraction rates (R_{MODEL}) and optimal well locations. Hydraulic conductivity is a crucial factor: by overpredicting the factor, the model is likely to underpredict the extent of groundwater capture, and R_{DESIGN} will be higher than necessary; if hydraulic conductivity is underpredicted, R_{DESIGN} is likely to be less than needed to achieve the full required capture zone.

Following P&T system design and construction, remediators must evaluate the capture zone to determine if the system captures contaminants under diverse conditions, which can include varying hydraulic conditions, high or low precipitation, seasonal impact and changes in off-site pumping rates.

Five sites studied

Design consultants studied five P&T systems. For each system they used a groundwater model to derive extraction rates. Consultants did not clearly identify the actual rates that applied to potentiometric surfaces generated during capture-zone evaluations. None of the consultants reported actual extraction rates in their routine monitoring reports, except when requested by the U.S. EPA. None of the systems produced R_{DESIGN} .

For three of the systems, the U.S. EPA required actions to improve average extraction rates and increase groundwater capture. Consequently, additional investigations, implementation of corrective measures and additional government oversight were necessitated.

System overhauls and improvements

The first two P&T systems each used six extraction wells to capture groundwater from a shallow aquifer. Filtration, an ultraviolet light/hydrogen peroxide system, and air stripper (for P&T 1 only) treated contaminated groundwater. Corrective actions undertaken to increase

extraction rates for these systems consisted of the following:

- improved reporting of extraction rates and shut-downs
- changing the pretreatment filtration media from 10-micron plastic to 3-micron cotton
- increasing the frequency of replacement of media to reduce the frequency of UV/HP tube cleaning cycle
- improving telecommunications to reduce avoidable shutdowns
- improving maintenance to remove desiccant from air lines, which provide power to extraction well pumps
- initiating an extraction-well maintenance program, including more frequent cleaning to prevent plugging
- initiating more frequent inspection and maintenance of pipes and pumps to prevent plugging
- cleaning the oil/water separator
- installing a higher discharge pressure pump prior to pretreatment filtration and prior to post-treatment filtration
- upgrading or replacing extraction well pumps
- reducing the frequency of the UV/HP tube-cleaning cycle
- discharging rather than recycling the UV/HP tube-cleaning water
- increasing the spare-parts inventory
- attempting/achieving more rapid operator response to downtimes
- increasing operation extraction rates to attempt to compensate for downtimes

P&T 3 consisted of a groundwater extraction well at the source area and two lines of blocking wells designed to contain source area groundwater in both a sand-and-

gravel and upper-sandstone aquifer. Groundwater from the source area extraction well and blocking wells was pumped to an air stripper with carbon off-gas treatment. Initial corrective actions to increase extraction rates and improve capture consisted of

- improving reporting of extraction rates and shut-downs
- upgrading the wet-well pumps
- instituting a regular well-cleaning program
- installing alarms to warn of unexpected downtimes

P&T 4 covered one acre and consisted of nine extraction wells designed to contain source area groundwater in the sand-and-gravel and upper-sandstone aquifer. An on-site air stripper with carbon off-gas treatment was used to treat contaminated groundwater. To make the system more efficient, remediators abandoned an unproductive extraction well and cleaned fouled extraction wells. Remediators plan to decrease flow resistance by installing a replacement discharge line.

The P&T 5 site achieved hydraulic capture although actual extraction rates are below design rates. Extraction rates were lower than design rates because of

- shutdowns
- filter cartridges requiring replacement
- backwash of carbon oxidizer
- flame failure on the catalytic oxidizer
- power interruptions
- leak repairs

Hydraulic conductivity impact

Boice (2002) reports that the extraction rates of P&T 1, P&T 2 and P&T 3 fell short of the design rate because of "underdesign of the P&T systems, or a lack of conservatism in the design combined with little capacity to expand." To evaluate this problem, system designers

examined hydraulic conductivity. In general, hydraulic conductivities used in the original design were much too low, so these values were reevaluated with pumping tests. Results are as follows:

- P&T 1 and 2 were inadequately designed, and additional expenses were incurred because capture zones needed to be reevaluated
- an additional extraction well was added to P&T 1
- P&T 2 may need to be expanded or revised
- re-evaluating hydraulic conductivities with sensitivity testing may have reduced expenses

At P&T 3 (a sand-and-gravel aquifer), slug tests indicated an unrealistically low hydraulic conductivity, which may be responsible for the lack of containment. More testing is required, and this may indicate a need to expand the extraction system. In addition the blocking well system may not be providing total containment based on low-level detections in downgradient monitoring and supply wells. At this site

- higher costs were incurred to pay for additional investigation
- additional monies may be required to pay for future design, construction and close government oversight
- other expenses included
 - installing and sampling eight new nests of deep monitoring wells in the blocking wells
 - costs for monthly meetings
 - extensive additional sampling
 - installation of five additional monitoring wells
 - costs for legal and technical support

In contrast, P&T 5 will require no further construction costs because the aquifer was thoroughly characterized, and hydraulic conductivity values were applied conservatively at the design stage. Previous improvements included adding two new extraction wells with

pumping rates of 200 gpm each.

Conclusions

Boice draws three lessons from this study. First, if aquifers are not sufficiently characterized, and pump-and-treat facilities are not designed with enough conservatism or flexibility to compensate for uncertainty in the aquifer's hydraulic conductivity, additional expenses may be incurred. Accurate characterization of the hydraulic conductivity is most critical. A phased approach to P&T system construction and hydraulic testing can increase the success of a facility design. Assumptions used to model capture zones should be rechecked if water-level measurements do not clearly indicate attainment of the required capture zones.

Second, results of this study indicated that operation and maintenance was a more difficult task than expected and warrants more attention. "Designers need to recognize that P&T systems have downtimes, and therefore, the uptime pumping needs to be sufficient to maintain capture. In general, design consultants need to think of achieving R_{MODEL} on an average basis rather than only when the P&T system is fully operating. Possible average flow reductions, due to normal operation and maintenance problems, can be assessed as part of a sensitivity analysis for the design model. The impact of various groundwater extraction rates on attainment of the required capture zone can also be evaluated" (Boice, 2002).

The third lesson is based on the idea of "actual operation of pump-and-treatment systems: groundwater extraction rates were not initially correlated to capture-zone evaluations, nor were they included in routine operating reports. This should be done by identifying $R_{AVERAGE}$ in capture-zone evaluations and in routine operating reports. $R_{AVERAGE}$ should be compared to R_{MODEL} to assess if design objectives are being met. $R_{AVERAGE}$ should be

included in routine operating reports to confirm maintenance of pumping rates" (Boice, 2002).

Reference

Boice, R.E., "Extraction Rate Problems Lead to Increased Costs at Pump-and-Treat Facilities: A Call to Improve Reporting of Rates," *Ground Water Monitoring and Remediation*, Spring 2002; <http://www.ngwa.org>.



An innovative horizontal/vertical well

The rupture of a 16-inch-diameter underground fuel pipeline in Bellingham, Washington, created a gasoline plume of approximately 230,000 gallons. Remedial options considered included capturing seeps on the bank slope, excavating an interceptor trench, installing a grout curtain barrier or a frozen ground barrier, or drilling a horizontal directional drilled (HDD) borehole coupled with a vertical recovery well. Remediators selected the last option, the horizontal/vertical wall couplet (Carlisle and others, 2002).

Most of the gasoline from the spill flowed overland into two creeks, one of which is a salmon-bearing creek that discharges into Puget Sound. Gasoline migrated in the subsurface along two major pathways:

- the utility backfill surrounding the city's large-diameter water supply-distribution piping
- near-vertical bedding planes and fractures in the underlying sedimentary bedrock

Permeable backfill and a fractured bedrock aided rapid migration. "Within two days of the release, free-phase gasoline (light nonaqueous phase liquid—LNAPL) was observed seeping from a pristine, unburned bank of Whatcom Creek more than 300 feet north of the release point. The LNAPL seeps emerged along a 50-foot-wide zone near the base of the steep stream channel

bank, approximately 1 to 4 feet above the water level in Whatcom Creek" (Carlisle and others, 2002).

Gasoline appeared as a slight sheen to a steady stream of liquid. Remediators initially used booms, sorbents and hydrophobic product skimmers to contain and remove gasoline, but a more proactive remedial action was selected for the following reasons:

- gasoline-generated hydrocarbon vapors in the stream valley created a potential fire hazard
- gasoline migrated to seep locations relatively quickly, indicating a rapid fluid flow velocity within the bedrock fractures; bedding and fractures in the local bedrock created a strong north-south bias for gasoline migration
- other nearby groundwater seeps into Whatcom Creek could begin carrying gasoline
- the seeps occurred within a natural setting upstream of a popular public swimming destination
- the containment boom and skimmer system required 24-hour-a-day maintenance to collect LNAPL and prevent overflow

Remediation system

Any remedial system selected would need to

- influence the entire width of the observed gasoline seepage
- intercept thin, near-vertical fractures
- be constructed without risk of damage to existing city utilities
- minimize negative aesthetics during construction and operation

In addition the system would need to address

- topographic complexities
- proximity of a large-diameter water-supply distribution piping
- local geology (for instance, seepage along the south bank of Whatcom Creek was found to be controlled by the north-south orientation of bedding planes and fracture sets)

The remedial options considered were

- direct capture of seeps on the bank slope
- an excavated interceptor trench
- a grout curtain barrier
- a frozen ground barrier
- a horizontal directional drilled (HDD) borehole coupled with a vertical recovery well

Remediators selected the last option, the horizontal/vertical wall couplet, for several reasons:

- constructability, considering logistical constraints
- ability to modify groundwater levels and minimize the potential for new gasoline seeps to emerge in nearby areas
- opportunity to apply vapor extraction simultaneously with groundwater pumping
- favorable cost benefits

Well construction and system design

Drillers completed the 420-foot-long HDD borehole in six days. Because workers drilled through material saturated with gasoline, air monitoring was conducted. Biodegradable fluid used was selected because of its ability to minimize the potential for impacting fracture permeability. Workers back-reamed the horizontal well to a completed diameter of 8 inches and used water-jetting techniques to develop it. Next they drilled the vertical boring (RW-1) directly above the midpoint of the horizontal well, and it penetrated the horizontal well at 36 feet below ground surface. This vertical well was drilled in six days. Appearance of a portion of the horizontal well's PVC sleeve indicated successful penetration by the HDD.

System designers installed a submersible pneumatic total-fluids recovery pump; risk of seepage occurring on the bank of the nearby creek (Whatcom Creek) made it critical to maintain the target pumping level in this well.

The horizontal/vertical well system was operational approximately two months after the initial pipeline release. Recovery rates for the gasoline-contaminated groundwater from the horizontal/vertical well couplet system were generally in the range of 0.5 gallon/minute (gpm), and have been up to 1 gpm following moderate to high precipitation events (0.5 inch or more in 24 hours). Continued operation of the pump in RW-1 led northward-migrating water and gasoline into the horizontal well drain, thereby bypassing Whatcom Creek.

To remove gasoline vapors and treat contaminants in the unsaturated zone, workers applied vacuum pressure (approximately 10 inches of water column) on RW-1.

"The horizontal/vertical well couplet was incorporated into a larger-scale remediation system for treatment of gasoline-contaminated soil in the release area. The large-scale system involved five vertical dewatering wells and 19 soil vapor extraction wells. Infrastructure (wells and conveyance piping) for air sparging also was constructed at the site for possible future use" (Carlisle and others, 2002).

Maintenance

Carlisle and others (2002) report that "reduced performance of the horizontal/vertical well system was observed to correspond with the accumulation of two or more feet of sediment in the base of RW-1. Therefore, a maintenance program was developed to provide periodic removal of sediment from the horizontal/vertical well system. An air-entrained suction technique using a vacuum truck has been successful in cleaning out RW-1 to its base at 36 feet below ground surface. Removal of sediment approximately monthly was necessary during the first three months of system operation; subsequently, sediment has been removed about once every three months. Less than 0.5 cubic yard of sediment is recovered during a typical cleaning event, along with 100 gallons or more of water."

Summary

Carlisle and others (2002) assert that their system was successful because it

- accounted for geologic conditions such as crosscutting (or transecting) zones of bedding plane weakness and fracture sets
- was practical, and costs were considerably lower than other methods evaluated
- was functional, adaptable, reliable and had a low visual impact

The system was installed in August 1999. Since January 2000, 2 million gallons of gasoline-contaminated groundwater, 325 gallons of LNAPL and 460 gallons of gasoline in dissolved and vapor phase have been collected.

Reference

Carlisle, D., Cook, D.A. and J.A. Miller, "Successful Use of an Innovative Horizontal/Vertical Well Couplet in Fractured Bedrock to Intercept a Mobile Gasoline Plume," *Ground Water Monitoring and Remediation*, Spring 2002; <http://www.ngwa.org>.

MTBE degradation and intermediates formed by Fenton's reagent

MTBE degradation using hydroxyl radicals (OH•) has been successful in the lab. Examples of MTBE degradation by hydroxyl radicals include the following

- MTBE oxidation by OH• generated from ultraviolet light and hydrogen peroxide; removal of 99.99 percent of the MTBE was achieved in 2 hours; intermediates included TBF (tert-butyl formate) and TBA (tert-butyl alcohol)

- MTBE degradation by sonolytic destruction (forming the by-products listed above)
- MTBE chemical oxidation by Fenton's reagent (FR), creating intermediate by-products such as TBF, TBA, methyl acetate and acetone (for MTBE at low concentrations of 1-2 mg/l)

This article describes the intermediate by-products produced via Fenton's reaction, which include TBF, TBA, methyl acetate and acetone (Burbano and others, 2002).

Experimental setup

Researchers performed MTBE degradation experiments in borosilicate glass bottles using 3.5 liters of deionized water and initial concentrations of chemicals (for instance, MTBE, acetone) of about 2 mg/l. A single dose of Fenton's reagent was added at a molar ratio of FR:chemical compound of 10:1. Researchers used a gas chromatograph with a flame ionization detector (FID) and capillary column to take samples at 0, .5, 1, 2, 5, 10, 15, 20, 30 and 60 minutes. Detection limits were 0.1 µg/l.

Experimental results

Degradation results for MTBE, TBF, TBA, methyl acetate and acetone are described below.

MTBE. Burbano and others (2002) report that "Degradation of MTBE and intermediate by-product generation were fast and appreciable during the first 2-3 minutes of reaction, and later the rates decreased and finally diminished after 30 minutes. At 30 seconds, the extent of degradation reached 68.2 percent and after 1 hour, 97.9 percent. These results clearly show that MTBE is easily oxidized by OH• attack. ... TBF was found to be the most abundant intermediate, followed by acetone, and at a lower level, TBA and methyl acetate. Acetone

was the only intermediate that showed a continuous buildup throughout the reaction time. This indicates a lower reactivity with $\text{OH}\cdot$."

"The large amount of TBF generated suggests that the amount of FR was not sufficient to make the reaction progress significantly beyond the initial stages." Data—acetone concentration with time—indicate

- poor reactivity with $\text{OH}\cdot$.
- the occurrence of parallel pathways of its own generation from MTBE, TBA and TBF when attacked by $\text{OH}\cdot$.

Researchers also report that degradation of MTBE by $\text{OH}\cdot$ is a complex process. In addition, compounds such as acetic acid and formaldehyde are also produced.

TBF. Experiments beginning with TBF with the same FR dose showed a much lower extent of degradation compared to MTBE. "The reaction also generated observable amounts of TBA and acetone. The consumption of hydrogen peroxide and Fe^{2+} during TBF degradation showed trends similar to those obtained for MTBE degradation, since noticeable residual concentrations were detected. After one hour of reaction time, the concentrations of Fe^{2+} and hydrogen peroxide were in each case reduced to 21.8 and 8.9 percent of their initial values (from 12.31 to 2.69 mg/l, respectively)" (Burbano and others, 2002).

Researchers report three important observations:

- methyl acetate was not formed
- the amount of acetone formed was small compared with the amount of TBA; this indicates that acetone is mostly generated from direct $\text{OH}\cdot$ attack on MTBE
- the observed amount of TBA generated from a given initial concentration of TBF is considerable, suggesting that TBA from MTBE degradation comes predominantly, if not totally, from TBF

TBA. Within the first 30 seconds, TBA degraded by 53.7 percent and by 60 minutes, 91.1 percent. Methyl acetate was not formed. Hydrogen peroxide decayed to 11.4 percent of its original value, while Fe^{2+} decreased to 13.5 percent of its initial value.

The major stable intermediate product generated from the reaction of TBA and FR is acetone. "...the presence of acetone as the most abundant intermediate proves the hypothesis that at least three parallel reactions lead to its generation during MTBE degradation by FR: MTBE, TBF and TBA oxidation by $\text{OH}\cdot$ attack.

Methyl acetate. Degradation amount of this compound was low, 7.7 percent in the first 30 seconds, and only 29.8 percent in 60 minutes. Hydrogen peroxide decayed from 7.58 to 2.33 mg/l (30.8 percent of initial value). Fe^{2+} concentration decreased from 12.54 to 8.14 mg/l (64.9 percent of the initial value). Methyl acetate is generated in the early stages of $\text{OH}\cdot$ attack on the MTBE molecule.

Acetone. Degradation rate was lower than compounds with the tert-butyl group, but higher than methyl acetate. Degradation reached 24.8 percent in the first 30 seconds and 64.6 percent after 60 minutes. After one hour of reaction, the concentration of hydrogen peroxide diminished from 7.58 to 1.17 mg/l (15.4 percent of the initial value). Fe^{2+} concentration decreased from 12.68 to 6.87 mg/l or 54.2 percent of the initial value.

Conclusions

Researchers concluded that during MTBE degradation, acetone was the product of at least three parallel reactions:

- $\text{OH}\cdot$ attack on MTBE
- $\text{OH}\cdot$ attack on TBF
- $\text{OH}\cdot$ attack on TBA

In addition, researchers found the following:

- compounds of simple structures like methyl acetate and acetone were less susceptible to $\text{OH}\cdot$ attack (as generated by FR) than those that include the tert-butyl group (MTBE, TBA and TBF)
- the oxidation of Fe^{2+} to Fe^{3+} follows the trend of compound degradation: higher yields for the compounds containing the tert-butyl group and lower values for the compounds with primary carbons
- compounds with ramifications in their structure, such as the tert-butyl group, have a more highly reduced state (lower oxidation state) than those with simple chains and are more susceptible to degradation when exposed to $\text{OH}\cdot$.

Reference

Burbano, A.A., Dionysiou, D.D., Richardson, T.L. and M.T. Suidan, "Degradation of MTBE Intermediates Using Fenton's Reagent," *Journal of Environmental Engineering*, September 2002; <http://ojsps.aip.org/eeo>. See also U.S. EPA's Web site, <http://www.epa.gov/swrust1/mtbe/index.htm>, for more information on MTBE.



Natural attenuation rate clarifications: the devil's in the details

By Joseph Odencrantz, Richard A. Vogl, Mark D. Varljen and Anthony Silva¹

This article will demonstrate the importance of clarifying and isolating the processes involved in the detailed examination of groundwater natural attenuation processes. We'll examine and comment on some recent work in the industry. Our argument is quite simple: when rates and half-lives are put in tables, let's make sure they are accurate.

The following statement, taken from a recent publication, illustrates the problem: "A biodegradation rate derived from mass changes was calculated from changes in total dissolved BTEX mass over time; this was used in a solute transport model to estimate the cleanup time" (Suarez and Rifai, 2002, in Odencrantz and others, 2002). Let's be very careful with our interpretation of site data and even more careful with what we call the results of our quantifications. Changes in dissolved-phase mass that are calculated using chemical concentrations from monitoring wells are limited by monitoring network quality and do not represent any one process.

Natural attenuation

The term "natural attenuation" (NA) has been defined as "naturally occurring processes in soil and groundwater environments that act without human intervention to reduce the mass, toxicity, mobility, volume, or concentration of contaminants in those media" (Wiedmeier and others, 1999, in Odencrantz and others, 2002). In-situ NA processes include biodegradation, dispersion, dilution, adsorption, volatilization and chemical or biological stabilization or destruction of contaminants; e.g., biodegradation is only one process of NA.

In practice, unfortunately, natural attenuation is often used synonymously with intrinsic bioremediation, self remediation, natural restoration, passive bioremediation or intrinsic remediation. "Natural attenuation" is not remediation. *Reviewer's comment: As defined by the U.S. EPA, monitored natural attenuation is a remedy; however, simply demonstrating that natural attenuation processes are operating at a site is not the same as demonstrating that they will achieve the cleanup goal in a reasonable timeframe.*

Natural attenuation occurs to some degree at every site; however, site conditions can limit its effectiveness

as an interim or long-term solution. Natural attenuation does not necessarily imply contaminant removal. Furthermore, site-specific conditions that often limit natural attenuation effectiveness as a contaminant removal/destruction process are rarely properly evaluated. It is vital that we distinguish between destructive processes and dilution. *(Reviewer's comment: the U.S. EPA does accept dilution as a process contributing to monitored natural attenuation.)* To do this, it is first necessary to establish the types of biological processes that may be induced or monitored at a site.

Puff of smoke

A research project at Ontario, Canada, involved injecting MTBE into the Borden Aquifer—and then looking for it after eight years. Researchers found only 3 percent of the injected mass and concluded that 97 percent had biodegraded—simply because they did not find the mass. This is analogous to trying to find all the smoke from a puff of smoke released to the outdoor air seven hours after its release (assume dispersion in air is 10,000 times that in water, eight years is 70,080 hours). Finding all of this smoke is clearly impossible, yet when reviewing this work, few scientists seemed to consider that perhaps the researchers simply didn't find—or couldn't quantify—the dispersed contaminant. *Reviewer's comment: They looked very hard for that puff. If it were there to find, they would have found it. The groundwater does not move as fast as air, or as far in eight years as air moves in seven hours.*

The work was excellent with respect to quantifying the natural attenuation of a small, instantaneous amount of MTBE; however, it did not document biodegradation. There was no definitive proof—such as the presence of metabolic byproducts—presented that suggests that MTBE underwent biodegradation. *(Reviewer's com-*

ment: Recent laboratory work carried out in a Borden aquifer-like environment supports the contention that MTBE biodegradation may have occurred in the Borden aquifer (Schirmer and others, 2002).) Unfortunately both the consulting and regulating communities assume intrinsic decay, neglecting to recognize that natural attenuation of MTBE can occur, under the right set of circumstances and in the absence of biodegradation. The line between dispersion/dilution and biodegradation can be drawn if remediators can determine if concentration changes represent changes in the plume's mass, or if the plume has moved to other places in the aquifer.

Mass flux and dilution

A paper published in 2001 presented a framework by which dissolved-phase mass of groundwater constituents mixes with water extracted from production wells. The mass-flux mixing approach takes the mass from a groundwater plume and mixes it with the water from typically larger, deeper flows and formations.

According to the authors, the capture zones "are useful for illustrating contaminant dilution in continuously pumped supply wells." The capture zones are the regions of groundwater that are pumped into a production well as a function of time. When multiple plumes are heading toward a municipal well, "the larger pumping rates of many municipal supply wells may be sufficient to cause enough blending so that contaminant concentrations in extracted water remain relatively low." The authors used multilevel well fences downgradient from the plume to estimate contaminant mass flux leaving the site. Each of 49 probes used in this exercise sampled 22 square feet of aquifer perpendicular to plume flow direction. Despite the efforts of an extensive monitoring array, the data seem to indicate that even this elaborate monitoring approach was not adequate. Transects along sides and bottom contained significant contaminant concentrations, implying that only a portion

¹The reviewer's comments for this article were provided by a researcher from U.S. EPA.

of the plume was sampled.

The example yielded a mass flux of 31 grams of a compound per day after multiplying by the calculated specific discharge—Darcy velocity of 0.64 inches/year—and adding up each mass flux from the individual probe areas. If this mass flux were to enter a municipal supply well pumping one million gallons/day (694.4 gallons/minute), the resulting concentration after mixing would be 8.2 $\mu\text{g}/\text{l}$. Average concentration at the fence was approximately 20,000 $\mu\text{g}/\text{l}$. The net effect is to lower the concentration by approximately 2,500 times once the water is pumped from the aquifer by the municipal supply well.

Does this imply that to manage contaminant plumes, we are relying on end-user dilution? (*Reviewer's comment: This is commonly done when there is no other choice.*) What does this say about our sampling results if monitoring wells are sampled with high-volume, high-flow purging and sampling techniques, or if the monitoring wells are located in areas that may underestimate plume dimensions?

Decay, dispersion and misnomers

Monitored natural attenuation protocols generally involve the collection of biogeochemical data from groundwater monitoring wells. The data are correlated in time and space with the various chemicals of concern (COCs) to establish predominant biodegradation mechanisms.

In evaluating the size, behavior and mass of groundwater plumes, monitoring wells are sampled by a variety of techniques at fixed locations, which the protocol assumes. The protocol also assumes that the monitoring wells fully delineate the plume and that there is an adequate number of wells to calculate a plume mass every time the wells are sampled. Unfortunately this is not often the case in practice.

Under this assumption, can we really explain what the plume is doing (i.e., expanding, stable or shrinking) by examining the history of a gasoline compound's concentration at a well? Of course this depends largely on where the well is located, how the well was constructed (type, screen length, etc.) and how it was sampled (i.e. low-flow, traditional purge or no purge). If the concentration rises and drops over a two-year period, does this mean

- that the plume is shrinking?
- that the plume has moved past the well?
- or, that there is a change in flow direction?

To answer this question, we need to look at

- the conceptual site model
- changes in concentration at other wells
- changes in biogeochemical parameters

These considerations are intuitive, and most practicing professionals routinely use standard methods and state guidelines to work through such evaluations. When evaluating the dominant attenuation processes, obtaining representative data from monitoring wells is a critical first step for isolating NA processes. The importance of collecting representative data (as influenced by well location, construction and sampling protocols) cannot be underestimated.

Assume for the moment, however, that we have not only an adequate number of wells to fully delineate our plume, but that there are only nondestructive NA processes at work (i.e., advection, dispersion, sorption and volatilization) and we can predict them perfectly using models (another assumption that is never really achieved in practice). If sorption and volatilization were minimal, plume mass would remain virtually constant if we calculated it each time from well water concentra-

tion. We adjust small changes in mass using sorption and volatilization models. This approach has been used at a variety of research sites where several transects of multilevel monitoring wells were placed perpendicular to the groundwater flow direction. If we had the typical monitoring wells at a service station site, however, and the same exercise was performed, it would be nearly impossible to make a reasonable estimate of the plume mass with time.

Continuing with our example, consider applying the BIOSCREEN model to estimate NA. We have a well at 30 feet downgradient (near-field) and one at 300 feet downgradient (far-field). We prepare to run the model by methodically estimating all the independent variables, such as source concentration, hydraulic conductivity/gradient and longitudinal dispersivity. We run the model with no first-order decay and find both wells are off significantly.

Because we do not have lines of evidence of biological degradation, we use a first-order "decay" coefficient to match field results. The near-field well matches with a first-order decay rate of 0.2 year and the far-field well matches with a rate of 1.5 years. It might seem reasonable that more decay would exist near rather than away from the source. Using the decay coefficient in this manner assumes decay is a lumped parameter because it is not specific to a mechanism such as biodegradation. In this case, it is used to account for loss of mass in a general sense. Perhaps the loss mechanism is not necessarily decay and there is more dispersion in the system than initially estimated. We ran the model with 10 times the dispersion without first-order decay and found the model output matched the well data (Figure 1). This suggests that first-order decay may not be occurring at some sites.

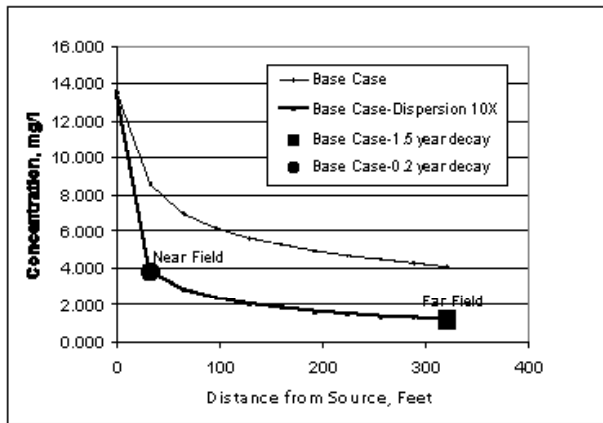


Figure 1. Increased dispersion and decay compared to base case.

Reviewer’s comment: BIOSCREEN in Run Center-Line estimates reductions in concentration, not total mass. There is a flux estimate in Run Array, but no one ever uses it. Run Array estimates the effect of dispersion on concentration. Most state regulators that I work with use BIOSCREEN to support their decision to run the model with no decay (no biological degradation) and with a reasonable value for dispersion. If concentrations are below action levels at the nearest receptor (with a safety factor), they spend their money to clean up another site. In other words, they don’t depend on biological degradation to meet cleanup goals.

We suggest that degradation rate, or half-life values, be scrutinized to

- clarify what processes the value represents
- establish how the value was determined
- make certain other processes such as dispersion are estimated correctly
- examine the available lines of evidence to substantiate if biodegradation rate is first-order

Unfortunately, the BIOSCREEN “Help” section encourages process mixing: “Modelers using the first-order

decay model typically use the first-order decay coefficient as a calibration parameter and adjust the decay coefficient until the model results match field data. With this approach, uncertainties in a number of parameters (e.g. dispersion, sorption, biodegradation) are lumped together in a single calibration parameter.”

Reviewer’s comment: BIOSCREEN was designed for BTEX with its biodegradation rates of 1/year; where we have laboratory evidence to back up the field rate for MTBE degradation, the rate is near or above 1/yr. At these rates, for a plume size typical of a UST release, at typical ground water flow velocities, the bulk attenuation rate is clearly dominated by the biological decay rate. Only when the biological decay rate is low (about 0.1/year) do dilution and dispersion become important. These low rates of biological decay are very difficult to estimate (with statistical significance) from field or laboratory data.

Buscheck and Alcantar approach

Buscheck and Alcantar (BA method) derived a steady-state solution to the advection-dispersion equation with first-order decay. Their approach assumes constituent concentration downgradient from a continuous source that can be translated as time away from the source by groundwater velocity. This coordinate transformation, along with a best-fit line through an exponential best-fit line, through the time vs. concentration plot, gives a decay rate estimate based on groundwater velocity, linear retardation factor and longitudinal dispersivity. Using the same parameters to generate the nonreactive profile (groundwater velocity—0.37 feet/day, retardation—1.0 and longitudinal dispersivity of 11.38 feet; see Figure 1), we determined the slope factor. Prior to this, we compared previously published work to the BA method and then calculated that a “decay half-life” of 2.29 years would be necessary to calibrate the base case profile (shown in Figure 1). Considering that this

profile was generated from a nonreactive case, we found any decay surprising. To explore the source of the discrepancy, we generated five more nonreactive profiles using BIOSCREEN at a range of different longitudinal dispersivities (see Figure 2). The curve matching the dispersivity of 113.8 feet is the base case (in Figure 1) times 10.

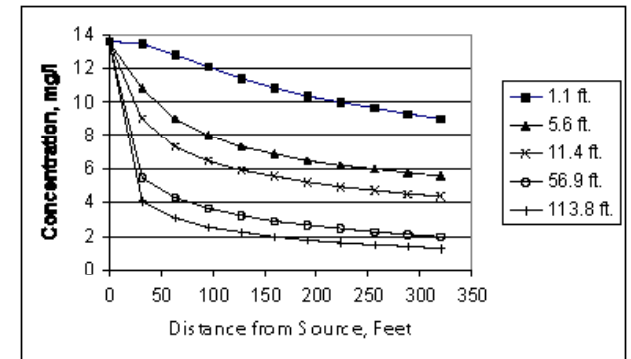


Figure 2. Concentration profiles at several dispersivities.

Next, we defined the BA decay half-life necessary to match the curves in Figure 2. Figure 3 is a plot of the decay half-life vs. dispersivity (using the BA model) to match a nonreactive case with different dispersivities. More decay is necessary to match the higher dispersivity values up to a dispersivity of 56 feet. At 85- and 133-feet dispersivities, the trend reverses. We ran four additional dispersivity values to find the minimum decay half-life.

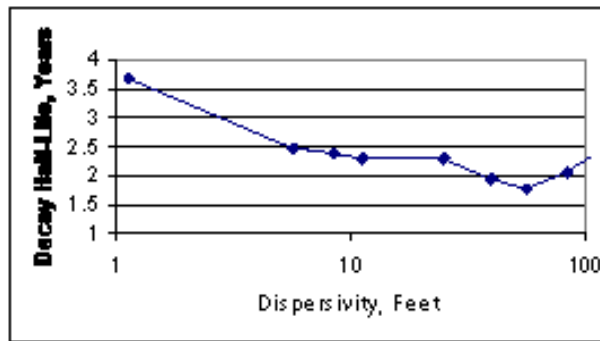


Figure 3. Buscheck-Alcantar decay half-life vs. dispersivity for a nonreactive case. The half-life at dispersivity of 113.8 feet (not shown on this figure) is 2.39 years.

Figure 3 illustrates the confusion that can arise by blindly using modeling approaches without being absolutely sure of the processes represented. Either the fundamental assumptions or the mathematical formulation associated with the BA approach are flawed.

Reviewer's comment: These authors are comparing a two-dimensional or three-dimensional projection of a particular release scenario (BIOSCREEN) to a one-dimensional analytical solution (BA). You would not expect them to agree. Actually, their agreement is not bad. With a 100-fold change in dispersivity, the extracted biodegradation term changes from 1.6 to 3.6. If you look at the error on the estimation of the slope of bulk attenuation used to calculate biodegradation in BA, the error bars are usually larger than this disparity.

Attenuation with distance in Run Array is fairly sensitive to source area and boundary width. Significant lateral dispersion in the first part of the flow path results in rapid attenuation. As the plume spreads laterally and vertically, concentration gradients are not as steep, and lateral and vertical dispersion have less influence on concentration. As you move away from the source, attenuation bulk rate by dispersion declines.

Decay or dispersion?

Profiles of the decay examples presented as "point-wise" results (Figure 1) were generated to determine if changes in dispersion might account for system behavior. Figure 4 indicates that a dispersivity of 350 feet (approximately 31 times larger than the base case dispersivity) matches the overall profile of the 0.2-year decay half-life. Recall that the simulation is a nonreactive case with a large amount of dispersion. The theory of dispersion is scale-dependent, and what may seem unrealistically large might be possible. MTBE plumes, for instance, can travel large distances. Figure 4 also shows that a 50-foot dispersivity (approximately four times larger than the base case dispersivity) successfully mimics the profile generated using the 1.5-year half-life.

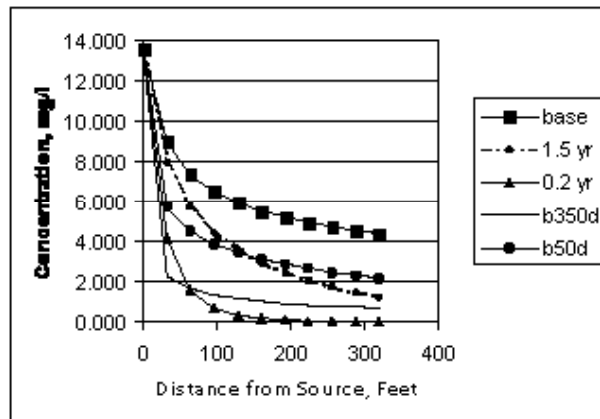


Figure 4. Decay and dispersion concentration profiles compared to the base case profile.

The final set of profiles we generated demonstrate the effect of groundwater velocity on decay and dispersion. These profiles (Figure 5) were generated about a new base case that consists of the previous base case with a groundwater velocity 10 times higher than the original case. The new base case dispersivity was also increased by a factor of 10 to 113.8 feet. The continuous source (first-type boundary) is distributing 10 times the mass into the system. Therefore, the 1.5- and 0.2-year decay half-lives have much smaller effects on the concentration distribution as compared to the previous base case. The next two profiles (b3500d and b500d) show the effects of 31 and 4 times the new base case dispersivity. Increased groundwater velocity and larger dispersivities have a much greater impact on plume dissipation because the two parameters are multiplicative, i.e., the effect of hydrodynamic dispersion is much larger than in the previous case. The last two profiles illustrate the effects of a more moderate amount of dispersion (b30d; 30 foot dispersivity) combined with a fairly significant decay half-life (b30d 0.2 yr; 0.2 years). This new decay is more pronounced on the higher velocity case with the smaller dispersivity (on the order of the longitudinal scale of the profiles, 30 feet or approximately 10 percent).

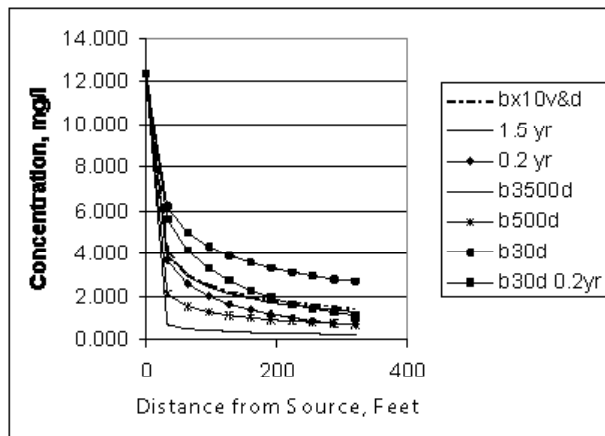


Figure 5. Profiles of base case with 10 times greater groundwater velocity and varying values of dispersion and decay.

Conclusions

Basing our conclusions on the examples presented and our own analyses, we caution others to recognize the limitations of relying on oversimplified analyses and lumping together discrete processes. Consider something as simple as groundwater velocity prediction, which relies on hydraulic conductivity, hydraulic gradient and effective porosity values. Hydraulic conductivity values, universally known and understood to varying degrees, still encompass much uncertainty. Although the uncertainty associated with hydraulic gradient and effective porosity is less, uncertainties multiply.

General suggestions for using degradation rates and half-lives include the following:

- be sure to clarify processes involved
- establish how they were determined
- ensure that other processes such as dispersion were estimated correctly
- if using a first-order biodegradation rate, examine available lines of evidence used in substantiation

References

Odenchantz, J., Vogl, R.A., Varljen, M.D. and A. Silva, "Natural Attenuation Rate Clarifications: The Devil's in the Details," paper presented at The American Petroleum Institute and National Ground Water Association Conference on Petroleum Hydrocarbons and Organic Chemicals in Groundwater: Prevention, Assessment and Remediation with Special Focus on Long-Term Site Management and Gasoline Oxygenates, Atlanta, Georgia, November 6-8, 2002. The original text, including references to other papers, is available at <http://www.tri-s.com>.

Schirmer, M., Butler, B.J., Church, C.D., Barker, J.F. and N. Nadarajah, "Laboratory Evidence of MTBE Biodegradation in Borden Aquifer Material," *Journal of Contaminant Hydrology*, Vol. 60, pp. 229-249, 2003; <http://www.elsevier.com/locate/jconhyd>.

UTTU thanks Dr. Joseph Odenchantz, jodencrantz@tri-s.com, for his help on this article.

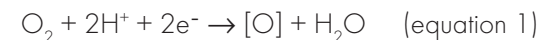
Note: The next issue of UTTU will contain articles on monitored natural attenuation and natural attenuation.

Treating MTBE and TAME with aluminum oxidation

During the last few years, researchers have developed several technologies to remediate MTBE. These have included advanced oxidation processes (AOPs) that use Fenton's reagent, ultraviolet (UV)/hydrogen peroxide, and ozone/hydrogen peroxide. Conventional techniques such as activated carbon adsorption and air stripping have not worked as well as AOPs in degrading MTBE. AOPs that produce highly reactive radical species (mainly hydroxyl radicals) require strong oxidants, such as hydrogen peroxide and ozone as precursors, and/or energy input (Lien and Zhang, 2002).

Dioxygen and processes

One oxidant used in AOP processes, albeit a weak one, is dioxygen (O_2). Studies have shown that the direct oxidation of oxygenates, such as tert-amyl methyl ether (TAME), by dioxygen occurs only at high temperatures. "In fact, even in the presence of strong oxidants such as ozone, the rate of the direct oxidation of MTBE is still very slow. However, the possibility of using dioxygen for the oxidation through a reductive activation of dioxygen involves the formation of reactive reduced oxygen species... Although the mechanisms for the dioxygen activation are complicated, this process can generally be expressed as equation 1 where [O] represents the reactive reduced oxygen species" (Lien and Zhang, 2002).



"The exact reactive reduced oxygen species involved in oxidation may depend on different experimental systems and conditions; nevertheless, the reduction of dioxygen to superoxide and subsequently to peroxide has been proposed (Figure 6). The utilization of these reactive reduced oxygen intermediates for oxygenate treatments seems impractical since they readily reduce to water, resulting in a very short lifetime. However, if a suitable reagent capable of stabilizing the reduced oxygen species can be developed, it might be used as a strong oxidant. For example, in the O_2/H_2 fuel cell system, the oxidation of alkanes and aromatic hydrocarbons through the reductive activation of dioxygen has been reported. Reduced oxygen species were generated at the cathode where they showed a finite lifetime in the presence of metal chlorides (e.g., $SmCl_3$) as catalysts" (Lien and Zhang, 2002).

requirement for molecule-to-molecule interaction. "For destruction of the contaminant to occur, the oxidant, catalyst, and contaminant must all come into direct contact. This will occur primarily on the main flowpaths. While some of the catalyst and oxidant will diffuse into the rest of the aquifer, the reactions in the other parts of the aquifer will be delayed and severely limited... the half-life of most oxidants is relatively short. The oxidants will self-destruct faster than they will diffuse."

It usually takes the oxidants two to eight weeks to self-destruct. According to field work studies, the aquifer will appear as clean, but contaminants will start to diffuse back into the main flowpaths. This increase in contaminant levels may result from a change in equilibrium stability of the contaminant; solubility equilibrium may be affected by destruction of natural humus material in the aquifer.

Reference

Lien, H.L. and W. Zhang, "Novel Bifunctional Aluminum for Oxidation of MTBE and TAME," *Journal of Environmental Engineering*, September 2002; <http://ojps.aip.org/eeo>.

Nyer, E.K., "Changing the Environment vs. Molecule-to-Molecule Reaction," *Ground Water Monitoring and Remediation*, Spring 2002; <http://www.ngwa.org>.



Research notes

Carbon Adsorption and Air-Stripping Removal of MTBE from River Water

Wilhelm, M.J., Adams, V.D., Curtis, J.G. and E.J. Middlebrooks, *Journal of Environmental Engineering*, September 2002; <http://ojps.aip.org/eeo>.

Researchers performed bench-scale studies using activated carbon and air stripping to evaluate the treatability of potential MTBE contamination in the nearby Truckee River (Nevada). Specifically, researchers sought to

- evaluate the proposed treatment alternatives for removing possible highly concentrated MTBE contamination from the Truckee River water at the bench-scale
- determine the feasibility of pilot- and full-scale use
- perform a cost analysis for each option
- recommend a course of action based on the experimental data and the results of the cost analysis

To achieve these ends, researchers first studied the carbon usage rate of three types of granular activated carbon (GAC) and three types of powdered carbon (PAC). Next they set up a bench-scale air-stripping system whose influent was MTBE-contaminated water. The values measured or calculated included

- MTBE by gas chromatography for each air/water ratio
- stripping factors
- mass transfer coefficients

Researchers found that the highest removal percentages corresponded to the highest air/water ratio.

Projected costs of treatment strategies are as follows. The activated carbon system, consisting of two towers in series along with the first load of 40,000 lbs of F-300 activated carbon, would cost \$150,000 (1998 dollars). "Assuming a treatment period of one day, 1.14×10^8 l/day (30 MGD), and based on the given information, 30 systems would be required and the capital cost would be \$4.5 million. The capital cost cannot be amortized because the equipment will only be used in the event of an emergency." Total operational costs were estimated at \$318,000.

Costs for the air-stripping system consist of the following:

- direct capital cost of the air stripper, \$273,500, multiplied by a design-contingency overhead factor (to account for potential deviations from the cost equation), for total direct capital costs of \$317,000
- indirect capital costs covering sitework, engineering design and construction, estimated at \$196,500
- total capital costs, \$513,500
- operational costs, \$18,000

The total cost of the treatment was estimated at \$4,818,000 or \$0.043/l

Researchers concluded that "Neither of the treatments examined in this study appears to be cost-effective for removing high concentrations of MTBE from a flow of 1.14×10^8 l/day (30 MGD)... In the event of an MTBE spill into the Truckee River, the best course of action would be to close the water treatment plant and wait for the contaminant to flow downstream and dissipate."

Comparison of Liquid and Gas-Phase Photooxidation of MTBE: Synthetic and Field Samples

Sahle-Demessie, E., Richardson, T., Almquist, C.B. and U. R. Pillai, *Journal of Environmental Engineering*, September 2002; <http://ojps.aip.org/eeo>

This study evaluated the effectiveness of liquid-phase oxidation using a falling film reactor and air stripping with downstream gas-phase photocatalytic oxidation to treat MTBE-contaminated water. Researchers used the liquid-phase falling film photooxidation reactor for degrading low concentrations of MTBE ($50 \mu\text{g/l}$ - $10 \mu\text{g/l}$) and a slurry concentration of 0.1 g/l TiO_2 , which showed high rates of degradation of MTBE and its byproducts, TBA and TBF. "Tests conducted using field samples of MTBE-contaminated groundwater showed that aromatic organic species (BTEX) were degraded in less than 150 minutes, but the MTBE degradation was deterred by the

presence of BTEX, dissolved metal and chloride ions. To obtain higher degradation rates, pretreatment of the contaminated water by pH adjustment, ion exchange or addition of other oxidants such as H₂O₂ might be necessary, at increased treatment costs."

Researchers state: "The integrated air stripping with photocatalysis air treatment process could be an alternative technology for high-flow-rate treatment of MTBE-contaminated waters... A longer residence time and increased amounts of oxygen could yield high removals but might be costly." They conclude that the two techniques require further evaluation to optimize the processes.

Comparison of Temporal Trends in VOC as Measured with PDB Samplers and Low-Flow Sampling Methods

Harte, P.T., *Ground Water Monitoring and Remediation*, Spring 2002; <http://www.ngwa.org>

Harte (2002) compared two sampling methods—passive diffusion bag (pdb) sampling and low-flow sampling—at two tetrachloroethylene (PCE) sites and found comparable results. Factors that impact on pdb sampler effectiveness include

- site hydrogeology
- well construction
- ability of VOC to diffuse through the sampler
- deployment time of the pdb sampler
- rate of exchange of water between the well and aquifer under unstressed conditions at the well

Harte studied pdb sampling at two observation wells outfitted with 5- and 10-foot screens. The aquifer was highly permeable, and hydraulic connection between the aquifer and wells allowed "good natural flushing of water between the well and aquifer." PCE concentrations varied by one order of magnitude in one well

and by one-third order of magnitude in the other well. According to Harte (2002), "The analysis of temporal trends in PCE concentration determined by the two sample methods indicates that pdb samplers adequately sampled the large variation in PCE concentrations at the site. The slopes of temporal trends in concentration were similar between the two methods, and the pdb sample concentration generally reflected the instantaneous concentration sampled by the low-flow method. Therefore the pdb samplers provide an appropriate sampling method for PCE at these wells."

Effect of BTEX on Degradation of MTBE and TBA by a Mixed Bacterial Consortium

Sedran, M.A., Pruden, A., Wilson, G.J., Sudian, M.T. and A.D. Venosa, *Journal of Environmental Engineering*, September 2002; <http://ojs.aip.org/eeo>

Researchers created a laboratory reactor consisting of salts, nutrients and a combination of degrading cultures to evaluate MTBE and BTEX degradation. Next, they set up batch studies using 90 ml of reactor effluent and 10 ml of biomass and spiked the microcosms with the following combinations:

- MTBE alone
- MTBE with BTEX
- TBA alone
- TBA with BTEX

The degrading cultures present consisted of Flavobacteria-Cytophaga, α - and β -Protobacteria and gram positive bacteria.

From these studies, researchers observed the following:

- BTEX did not have a noticeable effect on MTBE degradation but did have an effect on TBA degradation

- the MTBE and BTEX reactors outperformed the MTBE-only reactor in terms of TBA degradation rate and overall degradation rate
- TBA rates and overall rates were higher in the BTEX culture than in the MTBE-only culture at all initial concentration levels
- MTBE rates were higher in the MTBE-only culture than in the BTEX culture
- as with other studies, exceptionally high biomass concentration is required for efficient MTBE degradation

Sedran and others (2002) concluded the following:

- BTEX may assist in the development of biomass for overall MTBE degradation
- microbes prefer BTEX over MTBE or TBA; therefore, BTEX is likely to exhaust the available DO (dissolved oxygen) native to a contaminated aquifer
- "In this present study, MTBE degraded without a lag phase and at the same rate regardless of the presence of BTEX ... this is most likely due to the diversity of culture used in the study ... these diverse organisms apparently can overcome the metabolic inhibition observed in pure culture."
- in batch studies, BTEX degraded orders of magnitude faster than MTBE or TBA

Researchers further recommended that "A mixed culture rather than a pure culture will be the most effective strategy whether treatment is in-situ or ex-situ." They also suggest that ex-situ treatment is probably the best option for MTBE and BTEX co-contamination.

Tracer Tests and Image Analysis of Biological Clogging in a Two-Dimensional Sandbox Experiment

Kildsgaard, J. and P. Engesgaard, *Ground Water Monitoring and Remediation*, Spring 2002; <http://www.ngwa.org>.

Researchers devised some fairly sophisticated column experiments to demonstrate the effects of biological clogging. They then developed "tracer and image-analysis techniques that permitted a visual identification of the progress of biological clogging and that could give tracer concentration maps needed for calibrating numerical simulators. This kind of information increases understanding of dynamic clogging processes and complements or even replaces hydraulic head measurements. Such information is needed for better quantification of effects of biological growth on the hydraulic properties of a porous medium" (Kildsgaard and Engesgaard, 2002).

The testing device researchers created was a 0.3 x 0.44 x 0.01 m Plexiglas sandbox with an inlet chamber at the base and an outlet (overflow) valve at the top. A manometer tube was also at the chamber's base. Nutrients added were acetate (electron donor) and nitrate (electron acceptor). "The sandbox experiments, therefore, mimicked a typical bioremediation situation, in which an extra primary source of carbon plus an electron acceptor are continuously supplied to the contaminated source area, where bacterial growth had occurred because of breakdown of a secondary contaminant source of carbon" (Kildsgaard and Engesgaard, 2002).

Researchers took the precaution of boiling the water that would be added to the sandbox to ensure it was free of oxygen, other dissolved gases and bacteria. The electron acceptor (nitrate), the substrate and a dye tracer were injected at the injection point. Microorganisms were obtained from a waste water treatment plant

(post-denitrification step). Addition of nitrate produced N_2 gas and gas bubbles, which would reduce hydraulic conductivity.

Researchers gathered data using a dye tracer (Brilliant Blue) and a digital camera. They found that the Brilliant Blue plume was not "entirely symmetric, indicating that packing the sandbox could have produced small-scale variability in the hydraulic conductivity." Still, transverse dispersion in the unclogged sandbox was very small. Researchers suggested that "clogging resulted in a finger-like spreading of the tracer around the main clogged area. Fingers were asymmetric and their dominant direction changed over time... The two fingers were maintained as two distinct plumes after the area clogged up to the time of stopping the injection. Some slow movement of the tracer into the area clogged also took place." The asymmetric biomass distribution may have resulted from

- inhomogeneities in the sand
- initial solid biomass distribution
- random detachment of the bacteria

Researchers concluded the following:

- the hydraulic gradient over the whole sandbox decreased very little
- the tracer experiment revealed a complex flow pattern with localized clogging effects
- to avoid or predict clogging, characterizing the transport behavior of nutrients and biomass in a porous system is extremely important



Information sources

U.S. EPA publications and information

Elements for Effective Management of Operating Pump-and-Treat Systems (EPA 542-R-02-009) can be downloaded from <http://clu-in.org/techpubs.htm>.

Recent videos/presentations from CLU-IN studio (<http://clu-in.org/studio>) include

- In-Situ Treatment of Groundwater Contaminated with Non-Aqueous Phase Liquid Contamination: Fundamentals and Case Studies
- See also the videos in the archives

The New Triad Section on CLU-IN is available at <http://clu-in.org/triad/>.

Other Web sites and documents

Contaminated Sediments in Superfund, <http://www.epa.gov/superfund/resources/sediment>

Documents from the European Union-Sponsored Contaminated Land Rehabilitation Network for Environmental Technologies in Europe, CLARINET, <http://www.clarinet.at/>, include

- Sustainable Management of Contaminated Land: An Overview
- Brownfields and Redevelopment of Urban Areas
- Remediation and Redevelopment of Urban Areas
- Review of Decision Support Tools for Contaminated Land Management and Their Use in Europe

Draft Guidance for Evaluating the Vapor Intrusion to Indoor Air Pathway from Groundwater and Soils (RCRA-2002-033), <http://www.epa.gov/correctiveaction/eis/vapor.htm>.

Environmental Monitoring and Measurement Advisor, EMMA, an expert system that helps users with critical questions concerning environmental monitoring, <http://www.EMMA-ExpertSystem.com>.

Groundwater Sensitivity Toolkit, designed to help managers and regulators prioritize groundwater sensitivity to a release, <http://groundwater.api.org/toolkit>.

Sediments listserv: send a blank note to mark.hodges@gtri.gatech.edu.

Technology Evaluation Report: Engineered Bioremediation (TE-02-03), http://groundwatercontrol.info/org/pdf/E_bio.pdf.

The National Environmental Methods Index (NEMI), database on environmental monitoring methods, <http://www.nemi.gov>.

UTTU obtained this information from *Ground Water Monitoring and Review* (<http://www.ngwa.org>), *Environmental Science & Technology* (<http://www.pubs.acs.org/>), and *TechDirect* (<http://clu-in.org/techpubs.htm>). UTTU thanks editors and writers for allowing us to reprint this information.