

DEMONSTRATING REMEDIATION BY NATURAL ATTENUATION USING NUMERICAL GROUND WATER MODELS AND ANNUAL GROUND WATER SAMPLING

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ABSTRACT

Activities at a former fire training area at Westover Air Reserve Base (ARB) in Massachusetts resulted in contamination of shallow soils and ground water with a mixture of fuel hydrocarbons and chlorinated aliphatic hydrocarbons (CAHs). Extensive hydrogeologic and geochemical data were collected in May 1995 and in July 1996. Ground water samples were analyzed to assess the potential for natural attenuation of benzene, toluene, ethylbenzene, and xylenes (BTEX), trichlorethene (TCE), dichloroethene (DCE), and vinyl chloride (VC). A numerical ground water model calibrated using hydrogeologic and geochemical data collected in 1995 was constructed to estimate the fate and transport of the dissolved BTEX compounds. Data collected during the second sampling round was used to assess the accuracy of model predictions and to confirm the effectiveness of natural attenuation processes.

Both rounds of data suggest that biodegradation of fuel hydrocarbons and CAHs is occurring. Data suggest that BTEX compounds are degrading through aerobic respiration and the anaerobic processes of ferric iron reduction, denitrification, sulfate reduction, and methanogenesis. A solute fate and transport model predicted that BTEX contaminant levels would increase over a 5-year period due to leaching of contaminants from soils into ground water. However, site data collected in 1996 indicate decreasing BTEX concentrations throughout the plume. Calculated decay rates at the site suggest approximately 36 to 54 percent of the dissolved BTEX is biodegraded per year. In addition, the decreasing ratio of TCE to the daughter product DCE suggests that TCE in the ground water is being degraded to DCE through reductive dechlorination. This process is advantageous in that BTEX compounds are oxidized while TCE is reduced to a compound that can be directly biodegraded to carbon dioxide and water.

SITE DESCRIPTION

Westover ARB is located in south-central Massachusetts near the Connecticut River. From 1964 to 1986, a former fire training area was used for training exercises in which fuels and solvents were spread over a replica of an aircraft fuselage and ignited. The volume of petroleum products and spent solvents used in the training exercises prior to 1974 is unknown. From 1974 to 1986, approximately 125 gallons of JP-4 jet fuel were used for each monthly training event. An underground JP-4 fuel storage tank was located northwest of the burn area, but was removed in 1986. An underground distribution pipe that transported fuel from the storage tank to the burn

area was abandoned in place. During tank removal, fuel remaining in the storage tank is believed to have discharged to the subsurface through the buried pipe.

PHYSICAL SETTING

The regional hydrogeology of the Westover ARB area consists of three major hydrogeologic units. An aquitard composed of lacustrine deposits and glacial till separates a shallow deltaic outwash aquifer from an underlying Triassic bedrock aquifer. Both aquifers are used to a limited extent for industrial, municipal, and domestic purposes. Base-wide hydraulic conductivities in the shallow aquifer average 13 feet per day (ft/day) and range from 2.2 to 33 ft/day. Depth to ground water is approximately 5 to 7 feet below ground surface (bgs) across the majority of the site. Across the northern and western portions of the site, ground water flow is to the southeast with an average gradient of 0.0018 foot per foot (ft/ft). An apparent convergent ground water divide, shown on Figure 1, is present just south of the main burn pit.

The convergent ground water divide appears to result from two interacting hydrogeologic conditions. The hydraulic conductivity data suggest that alluvial sand and gravel with a higher permeability than surrounding sediments is present in the vicinity of the convergent divide. Of the five hydraulic conductivities estimated for shallow ground water at the site, the two highest were measured at wells within and at the head of the convergent divide. The convergent divide is also thought to result from the interaction between regional and local ground water flow patterns. A southeastern regional flow direction has been observed for the northern portion of the base. This flow pattern is present over the northern and central portions of the site, in areas characterized by a general lack of topographic relief. The land surface rises to the west and south of the site. The ground water contour map (Figure 1) suggests that ground water from these topographically higher areas is flowing north and east into the topographically lower area of the site. The interaction of the southeastern and northeastern flows contribute to emphasize the preferential ground water flow through the zone of elevated hydraulic conductivity that trends to the east through the site.

Evidence suggests that vertical flow gradients within the shallow aquifer vary across the site. Three monitoring well clusters included shallow wells screened across the water table and deep wells screened at least 70 feet below the water table. Vertical gradients were computed at 0.024 ft/ft (downward) between wells CF-1 and CF-1A and 0.0043 ft/ft (upward) between CF-6 and CF-6A. The estimated vertical gradient was negligible between CF-2 and CF-2A. Dissolved contaminant concentrations suggest significant horizontal ground water movement through aquifer intervals from 5 to 15 feet bgs and 30 to 50 feet bgs. Given the migration of BTEX compounds to 30 to 50 feet bgs, a significant downward vertical gradient is believed to exist in the upper half of the surficial aquifer in the vicinity of the source area.

DISTRIBUTION AND EXTENT OF GROUND WATER CONTAMINATION

Historically, BTEX compounds have been detected to a maximum depth of 15 feet bgs in soil samples from the burn pit area, with the highest concentrations detected in soil from the vadose zone. In 1995, the maximum detected soil BTEX concentration was 440 milligrams per kilogram (mg/kg). With the exception of a soil sample at the outfall of the buried waste fuel pipe, soil samples collected outside of the burn area have not contained quantifiable levels of BTEX. CAHs have not been detected above detection limits in any soil samples.

The areal distribution of total dissolved BTEX and CAHs in ground water for July 1996 is presented on Figures 2 and 3. The main body of the plume is centered beneath the main burn pit with a secondary source located at the outfall of the abandoned and buried waste-fuel pipeline. The vertical extent of BTEX along the main axis of the plume parallel to the direction of ground water flow in 1995 and 1996 is presented on Figure 4. Vertically, the BTEX plume is split into two lobes separated by a silty sand unit with relatively low hydraulic conductivity.

The maximum observed total BTEX concentration in 1995 was 32,557 micrograms per liter ($\mu\text{g/L}$), detected in a ground water sample from monitoring point MP-4S. In 1996, the maximum BTEX concentration at the same location decreased to 26,125 $\mu\text{g/L}$. In 1995, the BTEX concentrations from the three shallow monitoring locations directly downgradient of MP-4S (i.e., MP-12S, CF-3, and CF-2A) had BTEX concentrations of 25,012, 6,266, and 3,020 $\mu\text{g/L}$, respectively. Samples collected from the same wells during July 1996 contained BTEX concentrations of 21,210, 4,047, and 2,006 $\mu\text{g/L}$, respectively.

The low horizontal ground water gradient at the site reduces the horizontal ground water velocity, and therefore BTEX compounds migrate or disperse in ground water both horizontally and vertically. The maximum detected 1995 BTEX concentration in the deeper saturated zone was 1,652 $\mu\text{g/L}$, in a sample from location MP-14D. In 1996, this concentration decreased to 1,254 $\mu\text{g/L}$. The intermediate-depth sample at the same location (MP-14M) contained dissolved BTEX concentrations of 324 $\mu\text{g/L}$ in 1995 and 173 $\mu\text{g/L}$ in 1996. The hydraulic conductivity of the silty sands in the intermediate zone is lower than that of the coarse sands of the deeper zone. The fact that the total BTEX concentration is higher in the deeper zone of the aquifer suggests that most of the contamination found in the deep zone migrates vertically through the silty sand upgradient of the CF-2/MP-14 cluster. The contamination then resumes a more horizontal flow path in the coarse sands of the deeper zone.

The May 1995 and July 1996 vertical distribution of CAHs along the axis of the plume parallel to the direction of ground water flow is presented on Figure 5. Vertical migration dominates in the chlorinated solvent plume. The maximum observed 1995 total chlorinated solvent concentration of 13,541 $\mu\text{g/L}$ was detected in shallow ground water near the suspected CAH source area (well CF-3). The second-highest 1995 chlorinated solvent concentration (708.3 $\mu\text{g/L}$) was observed in the sample from MP-14D, in the deep zone. In 1996, the total chlorinated solvent concentration in ground water samples decreased to 2,098 $\mu\text{g/L}$ at well CF-3 and 584 $\mu\text{g/L}$ at monitoring point MP-14D. In ground water samples from well CF-3, TCE concentrations decreased from 12,800 $\mu\text{g/L}$ in 1995 to 1,660 $\mu\text{g/L}$ in 1996. Above MP-14D in the silty sand layer, the total chlorinated solvent concentration in 1995 was 112.5 $\mu\text{g/L}$ (MP-14M), while in the upper zone of the aquifer at CF-2A, no chlorinated solvents were detected. In 1996, only 60.7 $\mu\text{g/L}$ of total chlorinated solvents were detected in a sample from monitoring point MP-14M. Furthermore, during both sampling events CAHs were not detected in the samples from CF-2 (below MP-14D), suggesting that the solutes are not migrating into the fine sand beneath the coarse sand unit. Chlorinated solvents were not detected above quantitation limits at any of the furthest downgradient shallow and deep ground water monitoring locations.

On the basis of 1995 site data, TCE accounted for 93 percent of the total detected chlorinated solvent mass; cis-1,2-DCE accounted for 6.8 percent; and PCE, 1,1-DCE, trans-1,2-DCE, and vinyl chloride accounted for the remaining 0.2 percent. The fraction of TCE is slightly higher in

the source area (CF-3) and lower downgradient from the source area. In the samples from MP-14D and MP-14M, TCE accounted for 76 and 23 percent of the total detected chlorinated solvent concentrations, respectively. The transformation of TCE to cis-1,2-DCE was more advanced in the 1996 sampling event. In 1996, TCE accounted for approximately 72 percent of the total dissolved CAH mass and the cis-1,2-DCE percentage increased to 28 percent. In the 1996 ground water sample from location MP-14D, TCE and cis-1,2-DCE each accounted for 49 percent of the total CAHs, while ground water from monitoring point MP-14M had only 17 percent TCE and 83 percent cis-1,2-DCE.

GEOCHEMICAL INDICATORS OF BIODEGRADATION

The degradation of BTEX and CAH compounds in ground water is usually accomplished through biologically mediated oxidation/reduction reactions, where microorganisms obtain energy for cell production and maintenance through the transfer of electrons from electron donors to available electron acceptors (Wiedemeier *et. al*, 1995 and 1996) . Electron donors at the site include natural organic carbon, fuel hydrocarbon compounds, and the less-chlorinated solvents (e.g., cis-1,2-DCE and vinyl chloride). Naturally occurring electron acceptors at the site include DO, nitrate, ferric iron, sulfate, and carbon dioxide. In reactions where BTEX is used as an electron donor, the native electron acceptors are preferentially used in the listed order. Redox potentials observed at the site suggest that the core of the plume is characterized by highly-reducing conditions associated with sulfate reduction and methanogenesis. Moving outward from the core of the plume the redox potentials steadily increase until potentials indicative of aerobic respiration are observed along the margins of the plume. Intermediate regions of the plume are characterized by denitrification and iron reduction. In the highly-reducing zone at the core of the plume, conditions are favorable for the use of CAH compounds as electron acceptors, whereas the aerobic fringes favor the use of less-chlorinated solvents as electron donors in redox reactions. CAHs also can be degraded through cometabolic processes in which the CAH is fortuitously transformed by microbial enzymes produced by microorganisms for other purposes.

The 1995 and 1996 electron acceptor data indicate that biodegradation of fuel hydrocarbons in the shallow aquifer is occurring via aerobic oxidation, ferric iron reduction, denitrification, sulfate reduction, and methanogenesis. This is evidenced by strong correlation between areas with elevated BTEX concentrations and areas with depleted dissolved oxygen, nitrate, and sulfate concentrations, and increased ferrous iron and methane concentrations. Table 1 summarizes ground water analytical and geochemical data gathered during the 1995 and 1996 site investigations. The average background DO concentration in 1995 and 1996 ground water samples was 9.4 mg/L. Within the area characterized by substantially elevated BTEX and CAH concentrations, DO concentrations range from 1.8 mg/L to 0.12 mg/L. Nitrate/nitrite (as N) has been detected in site ground water at concentrations ranging from <0.05 mg/L within the plume to 5.6 mg/L at the plume boundaries. Ferrous iron concentrations are as low as <0.1 mg/L in background ground water samples and in the core of the plume are as high as 280 mg/L in 1995 and 45.3 mg/L in 1996. Sulfate concentrations at the site range from <0.1 mg/L in the center of the plume to a maximum concentration of 76.7 mg/L. Outside of the BTEX plume, the methane concentrations are below the analytical quantitation limit. The highest methane concentrations, detected in ground water samples from the center of the plume, were 4.3 mg/L in 1995 and 14.6 mg/L in 1996.

The process where CAH compounds are used as electron acceptor in biologically-mediated redox reactions is termed reductive dechlorination. During this process, a chlorine atom is removed and replaced with a hydrogen atom. As a result it is possible for complete sequential dechlorination from TCE to ethene to occur under optimal geochemical conditions. At this site, TCE is reductively dechlorinated under anaerobic and reducing conditions to cis-1,2-DCE and vinyl chloride. cis-1,2-DCE is a more common intermediate than trans-1,2-DCE, which is a more common component of manufactured solvents. This is further substantiated through a comparison of the concentrations of TCE to the daughter products cis-1,2-DCE and vinyl chloride. Within the source area, the molecular mass ratio of TCE to DCE in 1995 was approximately 13 to 1 and vinyl chloride was not detected. Approximately 150 feet further downgradient the TCE to DCE ratio was less than 2.5 to 1, with vinyl chloride detected at a concentration of 2.2 µg/L. Although some of this decreasing ratio of TCE to DCE is due to the different sorptive capacities of the two compounds and subsequent migration rates (i.e., the retarded contaminant velocity of DCE is approximately 25 percent faster than for TCE), both the substantial decrease in the molar ratio of TCE to DCE and the appearance of vinyl chloride suggest that reductive dechlorination of TCE is occurring along the axis of the plume. Furthermore, chloride concentrations range from a background concentration of <0.5 mg/L to an average maximum concentration of 140 mg/L within the source area, implying that chloride is being released during the degradation of CAH compounds. As expected, the zone of reductive dechlorination overlies the source area, where low redox potential, high BTEX concentrations, and methanogenic conditions encourage use of TCE and DCE as electron acceptors.

Downgradient from the reductive dechlorination zone, the higher dissolved oxygen concentrations on the plume fringes appear to facilitate aerobic respiration of the TCE daughter products, cis-1,2-DCE and vinyl chloride. Aerobic respiration appears to have the greatest impact on the fringes of the plume, where cis-1,2-DCE and vinyl chloride apparently undergo electron donor reactions with the more oxygenated ground water. From 1995 to 1996, the total dissolved mass of cis-1,2-DCE decreased eighteen percent and the only detected concentration of vinyl chloride at the site also decreased. This implies that downgradient from the source area (where cis-1,2-DCE is being continually produced through reductive dechlorination), cis-1,2-DCE and any available vinyl chloride are being reduced in electron donor reactions. cis-1,2-DCE and vinyl chloride are less susceptible to reductive dechlorination because of their lower oxidation states. If electron donor reactions were not occurring in the less oxidized downgradient ground water, the cis-1,2-DCE and vinyl chloride plume mass and extent would be expected to increase. However, the production of cis-1,2-DCE and vinyl chloride in the source area and the decrease in total dissolved mass of each compound suggests that the less chlorinated CAHs are serving as electron donors in biologically mediated reactions downgradient of the source area.

BIODEGRADATION RATE CONSTANTS

Apparent BTEX biodegradation rates were estimated using the method of Buscheck and Alcantar (1995). An easterly ground water flow path through wells MP-12S, CF-2A, and MP-15S was used for estimating a biodegradation rate. This flow path represents a ground water travel path from the anaerobic plume core to the more aerobic downgradient extents. An exponential fit to the data estimates a decay constant of 0.0015 day⁻¹ in 1995 and 1996. Using the 1995 and 1996 data (Tables 2 and 3), first order plume attenuation rates of 0.001 and

0.0023 day⁻¹ were estimated for BTEX and TCE, respectively, by averaging the rate at individual points calculated using the Buscheck and Alcantar (1995) Shrinking Plume Method.

BIOPLUME II MODEL PREDICTIONS

The results of a site-specific Bioplume II model suggested that under current conditions, approximately 60 years would be required for remediation by natural attenuation (RNA) to reduce dissolved benzene concentrations to below the Massachusetts benzene maximum contaminant level (MCL) of 5 µg/L. Because there is no accurate record on the volume and frequency of fuel and solvent spills at the site, several conservative assumptions were made until the modeled plume matched the observed 1995 BTEX plume. The model was calibrated using two pumping periods that simulated an increasing source area and strength for a 20-year period (1966-1986) to represent the start of fire training activities and the buildup of a residual source. A third pumping period simulated the end of fire training activities using a 5 percent per year source weathering term within the burn area. An injection source upgradient from the site at the waste fuel tank was included in the third pumping period to simulate the release of waste fuel during the 1986 abandonment of the waste fuel tank and pipe. A final pumping period simulated 5 percent per year source weathering at all injection sources until the 1995 sampling event. Following model calibration, a 5 percent per year source decay (weathering) rate was used to simulate the fate and transport of the dissolved BTEX plume until RNA was complete. A calibrated anaerobic biodegradation rate constant of 0.001 day⁻¹ was used in the Bioplume II model for this site.

This model predicts declining plume concentrations, with the plume reaching its maximum downgradient extent in approximately 30 years. After 10 years, the extent of the 1,000-µg/L contour remains stable; however, the maximum BTEX concentration decreases by 33 percent from approximately 33,190 µg/L to 22,000 µg/L. After 30 years of weathering (year 2025), the dissolved plume reaches the maximum extent, approximately 150 feet downgradient from the calibrated position. At the modeled maximum downgradient extent, the BTEX plume does not leave the site, nor does it impact any potential receptors. In addition, while the BTEX plume is at its maximum downgradient extent, the 100-µg/L and 1,000-µg/L isocontours near the center of the plume are both receding. After 30 simulation years, the maximum BTEX concentration has decreased to approximately 4,550 µg/L, or 14 percent of the calibrated maximum concentration. After 62 years of natural weathering, the model suggests that the ground water plume will have almost completely attenuated, with a maximum BTEX concentration of 35 µg/L in the source area. Further model simulation suggests that after 62 years, the plume is completely degraded. A second ground water model which simulated the effect of RNA combined with soil bioventing, vertical ground water circulation, and excavation of contaminated soils at the waste fuel pipe outfall suggested a total time for site remediation of 30 years.

DISCUSSION AND CONCLUSIONS

The 1995 and 1996 ground water analytical results imply that BTEX and CAHs are being biodegraded. However, the 1996 ground water sampling results suggest that RNA at the site appears to be proceeding more rapidly than predicted by the numerical model. Several conservative assumptions were made during initial ground water modeling because insufficient contaminant data were available prior to the 1995 investigation. Therefore, the ground water model was calibrated only on the basis of the 1995 soil and ground water data. Consequently, the existing ground water model may be overly conservative. The model however, does suggest a

worst-case contaminant fate and transport scenario which could be recalibrated. If the present geochemical data base were used to recalibrate the Bioplume II model, less conservative and more realistic assumptions regarding contaminant fate and transport processes could be made.

On the basis of the 1995 ground water data and numerical modeling results, a remedial alternative consisting of RNA with long term monitoring (LTM), soil bioventing in the burn area, vertical ground water recirculation within selected monitoring wells, and excavation of the contaminated soils near the waste fuel pipe outfall was recommended. The present-worth cost of the proposed remedial alternative was approximately \$400,000 with modeled remediation time of 30 years. The estimated present worth cost of only RNA with LTM was approximately \$300,000 for 60 years of estimated remediation. Both of the proposed remedial alternatives included maintaining institutional controls on soil and ground water use at the site, and both are protective of human health and the environment. The more expensive remedial alternative was recommended because of the reduced time required for remediation.

Upon review of the 1996 monitoring results, these recommendations would likely change. At this site and other similar sites where soil and ground water do not present an immediate risk to potential receptors, a ground water monitoring program could be implemented for a 1 to 2-year time period to conclusively demonstrate RNA to the public and regulatory community and better establish degradation rates and model calibration points. Then, if required, the data from LTM program could be used to develop a ground water model that will more accurately simulate future site conditions for a remedial alternative analysis.

The cost of the 1996 ground water sampling event was approximately 15 percent of the total cost of the initial 1995 soil and ground water investigation. At this site, the additional expense of the second sampling event would substantially reduce the total cost of a recommended remedial alternative because more accurate (and less overly conservative) fate and transport predictions could be made using 2 years of geochemical data. If a monitoring program can effectively demonstrate that RNA will remediate a site, a more favorable response to natural attenuation would be expected from the regulatory and public community and ground water modeling may not be necessary.

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Table 1 Summary of 1995 and 1996 Ground Water Analytical data

| Sample ID | Date | Total BTEX (µg/L) | TCE (µg/L) | cis-1,2-DCE (µg/L) | Vinyl Chloride (µg/L) | Methane (mg/L) | Ethene (mg/L) | Dissolved Oxygen (mg/L) | Nitrite- Nitrate (as N) (mg/L) | Ferrous Iron (mg/L) | Sulfate (mg/L) | Chloride (mg/L) | Total Organic Carbon (mg/L) |
|-----------|---------|-------------------|------------|--------------------|-----------------------|----------------|---------------|-------------------------|--------------------------------|---------------------|----------------|-----------------|-----------------------------|
| MP-4 S | May 95 | 32557 | ND | ND | ND | 0.073 | 0.005 | 0.52 | < 0.05 | 100 | 2.41 | 8.1 | 60.2 |
| | July 96 | 26125 | ND | ND | ND | 0.237 | 0.008 | 1.23 | <0.05 | 37.3 | <0.5 | 1.6 | 42.6 |
| MP-5 M | May 95 | 49.7 | 4.40 | BLQ | ND | 0.362 | NA | 0.17 | < 0.05 | 9 | 2.28 | 8.4 | 4.3 |
| | July 96 | 23.9 | 3.20 | BLQ | ND | 1.54 | ND | 0.25 | <0.05 | 5.3 | 2.45 | 6.1 | 2.7 |
| MP-12 S | May 95 | 25012 | 4 | ND | ND | 0.031 | 0.003 | 0.12 | < 0.05 | 20 | < 0.5 | 1.1 | 67.8 |
| | July 96 | 21210 | 7 | ND | ND | 0.073 | <0.003 | 0.17 | <0.05 | 7 | <0.5 | 1.5 | 61.8 |
| MP-14 M | May 95 | 324 | 26 | 86.70 | ND | 3.421 | BLQ | 0.21 | 0.09 | 280 | 7.62 | 146.0 | 53.3 |
| | July 96 | 173 | 10 | 50.60 | ND | 8.79 | ND | 0.2 | <0.05 | 40.5 | 11.1 | 84.6 | 28.5 |
| MP-14 D | May 95 | 1652 | 541.0 | 158.00 | 2.20 | 4.286 | NA | 0.28 | 0.09 | 280 | 0.86 | 150.0 | 94.4 |
| | July 96 | 1254 | 289.0 | 288.00 | 1.70 | 14.63 | <0.003 | 0.24 | 0.09 | 45.3 | <0.5 | 131.0 | 77.4 |
| MP-15 S | May 95 | 0.92 | ND | ND | ND | 0.002 | BLQ | 0.67 | 0.7 | 9 | 20 | 3.2 | 7.1 |
| | July 96 | ND | ND | ND | ND | BLQ | ND | 6.21 | 2.73 | <0.05 | 3.16 | 1.6 | 1.5 |
| MP-15 M | May 95 | ND | ND | ND | ND | 0.001 | NA | 0.43 | < 0.05 | 7.5 | < 0.5 | 2.9 | 1.6 |
| | July 96 | BLQ | ND | ND | ND | 0.021 | ND | 0.15 | <0.05 | 2.1 | <0.5 | 3.3 | 1.2 |
| MP-15 D | May 95 | 46.8 | BLQ | ND | ND | 0.046 | ND | 0.24 | < 0.05 | 9 | 2.28 | 9.8 | 2.0 |
| | July 96 | 20.8 | ND | ND | ND | 0.129 | ND | 0.43 | <0.05 | 6.2 | 3.83 | 9.8 | 3.3 |
| CF-2 | May 95 | ND | ND | ND | ND | BLQ | NA | 0.34 | < 0.05 | 4.5 | 13.2 | 1.1 | 0.8 |
| | July 96 | ND | ND | ND | ND | BLQ | ND | 0.42 | <0.05 | 4.7 | 10.6 | <0.5 | 0.3 |
| CF-2 A | May 95 | 3020 | ND | ND | ND | 0.305 | ND | 0.4 | 0.22 | 3 | < 0.5 | 0.7 | 52.5 |
| | July 96 | 2006 | ND | ND | ND | 0.557 | ND | 1.8 | 0.1 | 3.8 | <0.5 | <0.5 | 40.0 |
| CF-3 | May 95 | 6266 | 12800 | 732.0 | ND | 0.008 | NA | 0.12 | < 0.05 | 10 | 1.43 | 1.9 | 25.8 |
| | July 96 | 4047 | 1660 | 434.00 | ND | 0.028 | ND | 0.17 | <0.05 | 5.4 | <0.5 | <0.5 | 24.7 |
| CF-5 | May 95 | 3.71 | 1.50 | 7.60 | ND | 0.004 | ND | 0.73 | 0.34 | 5.5 | 24.6 | 1.1 | 3.9 |
| | July 96 | 10.90 | 94.60 | 35.00 | ND | 0.008 | ND | 2.68 | 0.41 | 0.4 | 16.6 | <0.5 | 4.4 |

Table 2 Calculation of BTEX Decay Rate Using Shrinking Plume Method

| Well | C(t) [1996] (µg/L) | C _i [1995] (µg/L) | Time Between Samples (days) | Decay Constant (k) (day ⁻¹) |
|---------|--------------------|------------------------------|-----------------------------|---|
| MP-4S | 26,125 | 32,557 | 425 | 0.0005 |
| MP-5M | 24 | 50 | 425 | 0.0017 |
| MP-12S | 21,210 | 25,012 | 425 | 0.0004 |
| MP-14M | 173 | 324 | 425 | 0.0015 |
| MP-14D | 1,254 | 1,652 | 425 | 0.0006 |
| MP-15D | 32 | 52.1 | 426 | 0.0011 |
| CF-2A | 2,006 | 3020 | 425 | 0.0010 |
| CF-3 | 4,047 | 6,266 | 425 | 0.0010 |
| Average | | | | 0.0010 |

Table 3 Calculation of TCE Decay Rate Using Shrinking Plume Method

| Well | C(t) [1996] (µg/L) | C _i [1995] (µg/L) | Time Between Samples (days) | Decay Constant (k) (day ⁻¹) |
|---------|--------------------|------------------------------|-----------------------------|---|
| MP-5M | 3 | 4 | 425 | 0.0007 |
| MP-14M | 10 | 26 | 425 | 0.0022 |
| MP-14D | 289 | 541 | 425 | 0.0015 |
| CF-3 | 1,660 | 12,800 | 425 | 0.0048 |
| Average | | | | 0.0023 |

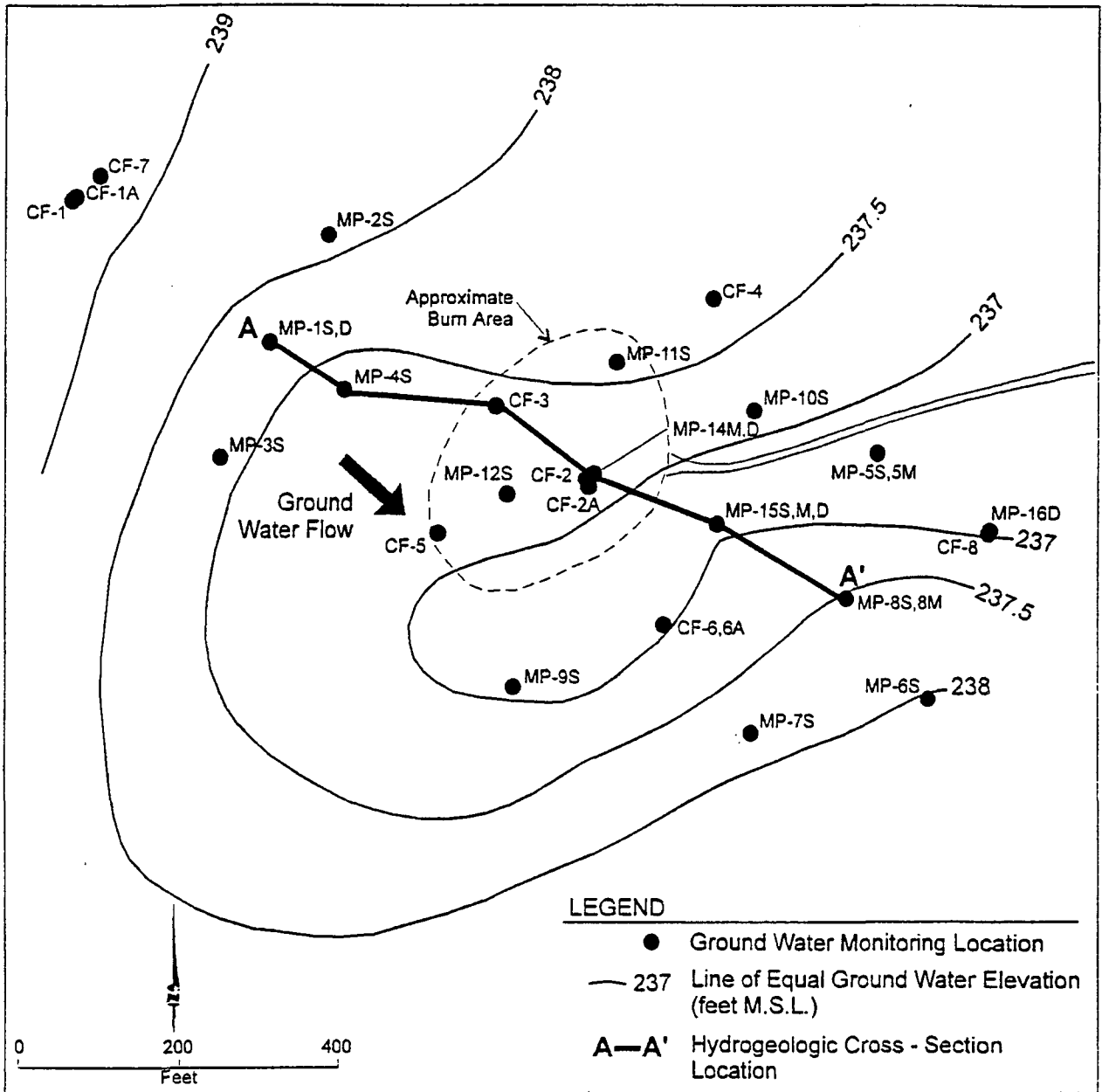


Figure 1. Site Layout and Ground Water Flow Map

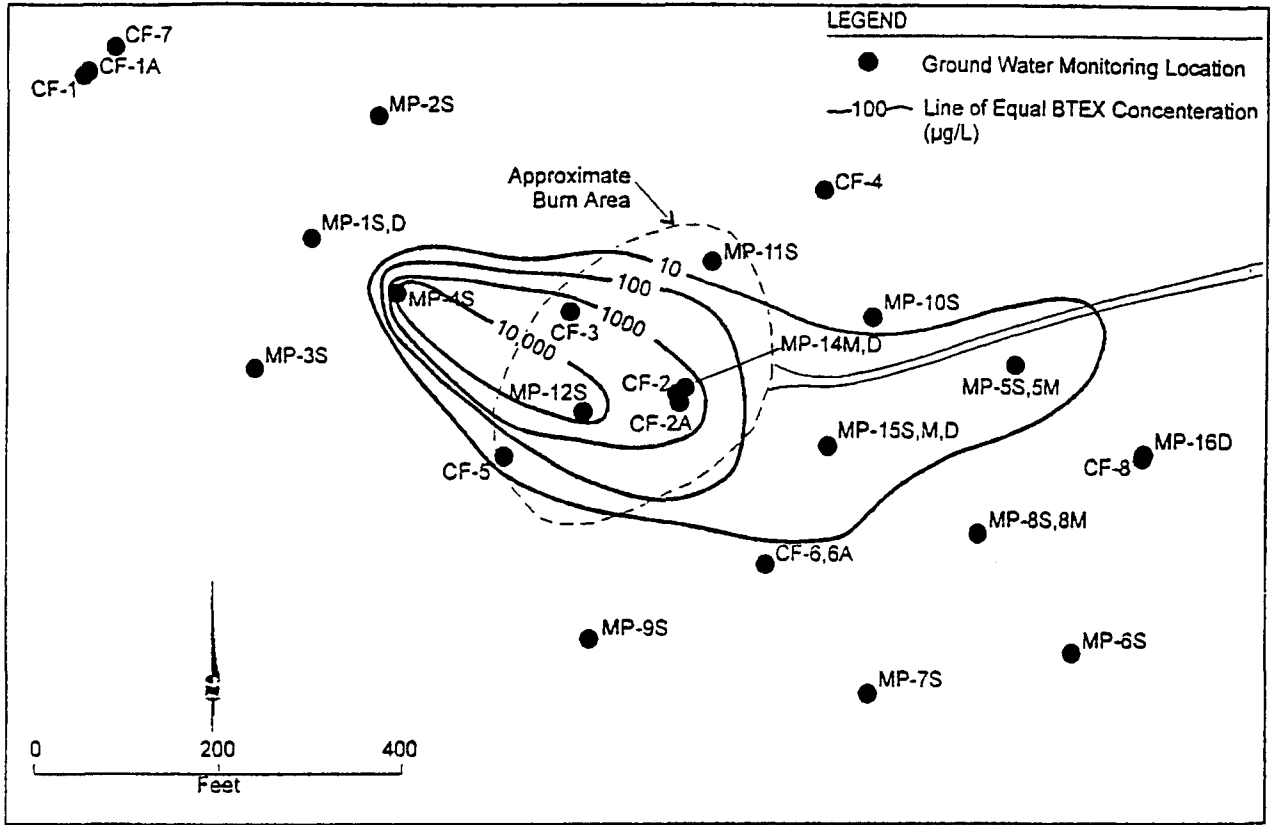


Figure 2. BTEX Isopleth Map for Ground Water, July 1996

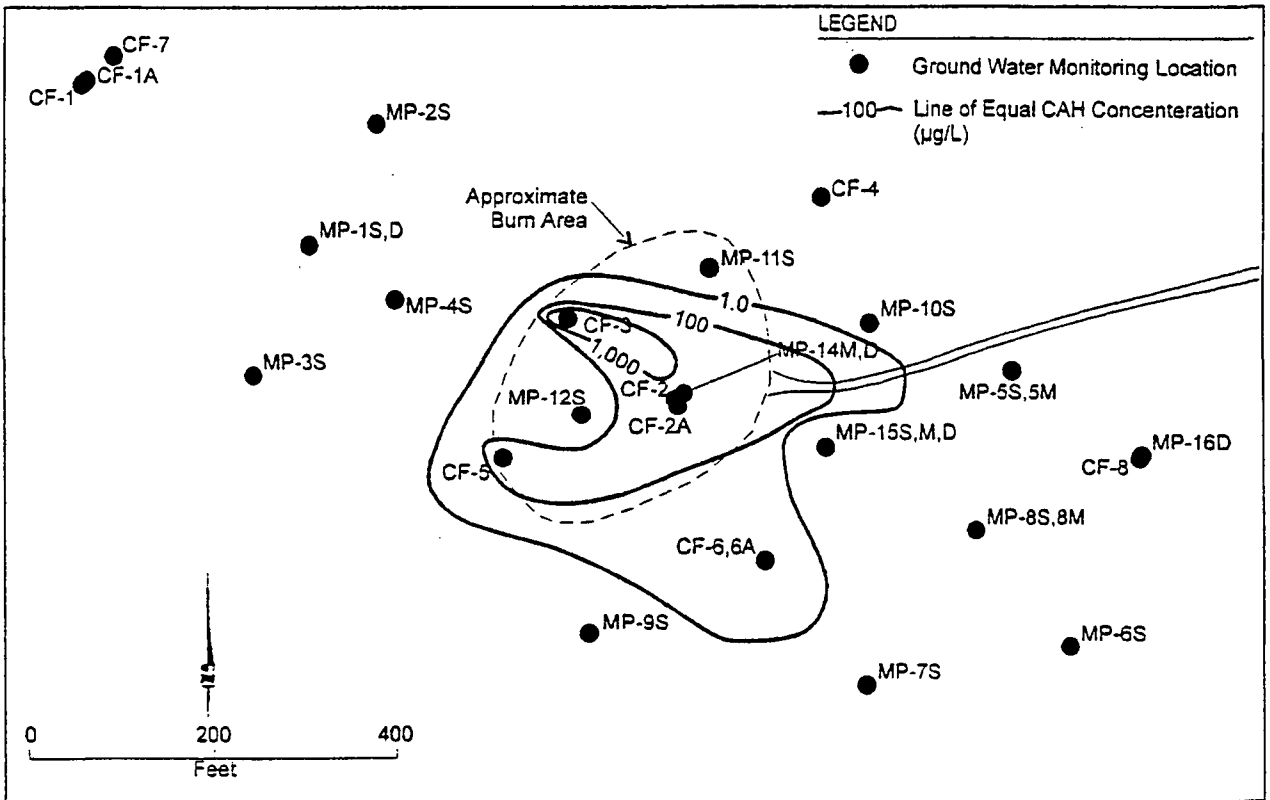


Figure 3. Total CAH Isopleth Map for Ground Water, July 1996

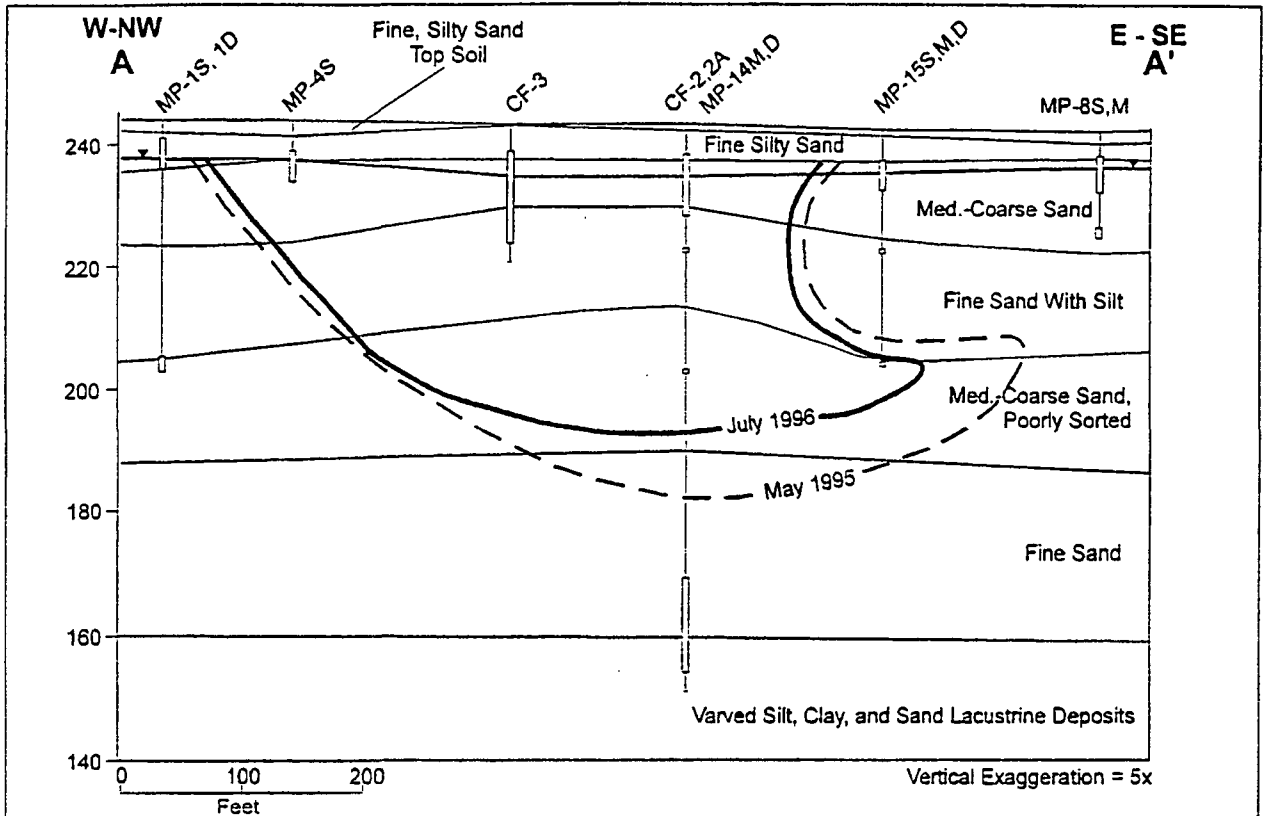


Figure 4. Vertical Extent of Dissolved BTEX, 1995-1996

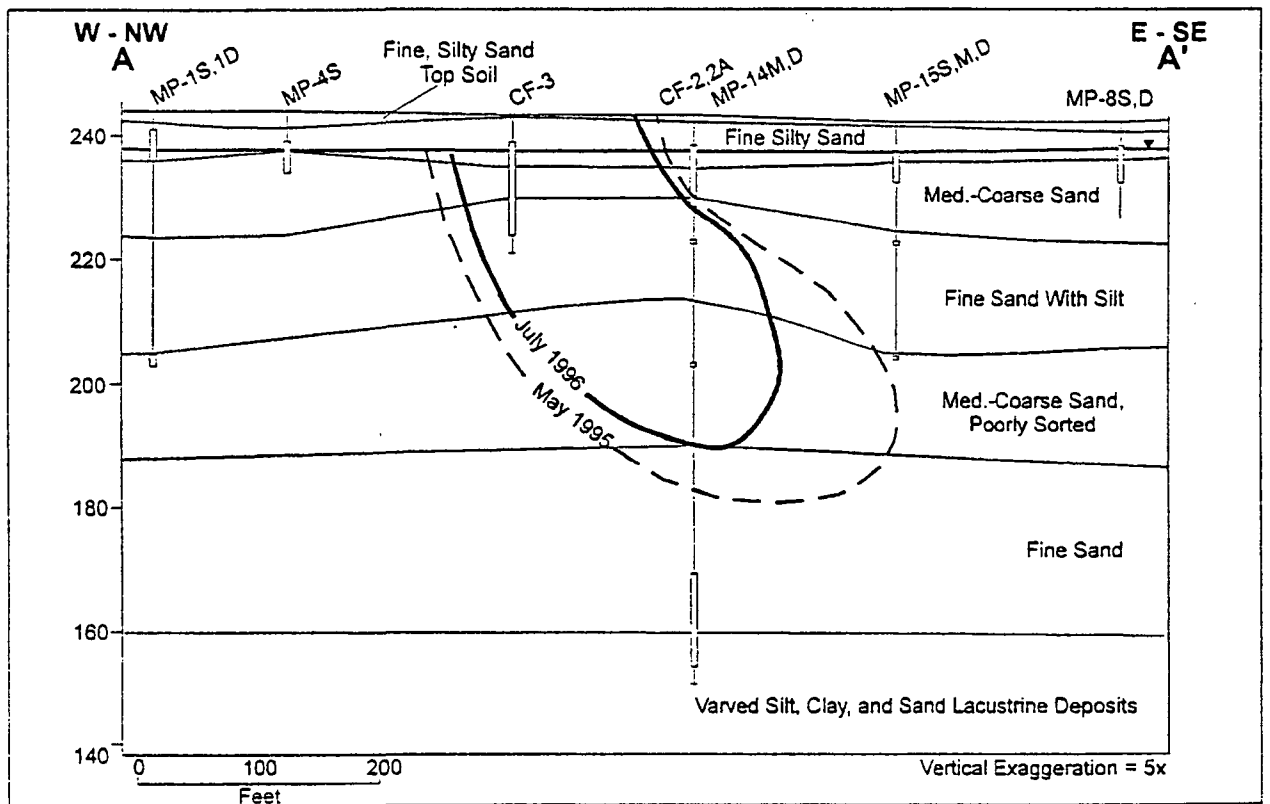


Figure 5. Vertical Extent of Dissolved CAHs, 1995-1996

BIOGRAPHICAL SKETCHES

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| 16. ABSTRACT Activities at a former fire training area at Westover Air Reserve Base (ARB) in Massachusetts resulted in contamination of shallow soils and ground water with a mixture of fuel hydrocarbons and chlorinated aliphatic hydrocarbons (CAHs). Extensive hydrogeologic and geochemical data were collected in May 1995 and in July 1996. Ground water samples were analyzed to assess the potential for natural attenuation of benzene, toluene, ethylbenzene, and xylenes (BTEX), trichloroethene (TCE), dichloroethene (DCE), and vinyl chloride (VC). A numerical ground water model calibrated using hydrogeologic and geochemical data collected in 1995 was constructed to estimate the fate and transport of the dissolved BTEX compounds. Data collected during the second sampling round was used to assess the accuracy of model predictions and to confirm the effectiveness of natural attenuation processes. Both rounds of data suggest that biodegradation of fuel hydrocarbons and CAHs is occurring. Data suggest that BTEX compounds are degrading through aerobic respiration and the anaerobic processes of ferric iron reduction, denitrification, sulfate reduction, and methanogenesis. A solute fate and transport model predicted that BTEX contaminant levels would increase over a 5-year period due to leaching of contaminants from soils into ground water. However, site data collected in 1996 indicate decreasing BTEX concentrations throughout the plume. Calculated decay rates at the site suggest approximately 36 to 54 percent of the dissolved BTEX is biodegraded per year. In addition, the decreasing ratio of TCE to the daughter product DCE suggests that TCE in the ground water is being degraded to DCE through reductive dechlorination. This process is advantageous in that BTEX compounds are oxidized while TCE is reduced to a compound that can be directly biodegraded to carbon dioxide and water. | | | |
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