

In cooperation with the U.S. Environmental Protection Agency

# Assessment of Potential for Natural Attenuation of Chlorinated Ethenes and Ethanes in Ground Water at a Petrochemical Reclamation Site, Harris County, Texas

Water-Resources Investigations Report 00-4121



U.S. Department of the Interior  
U.S. Geological Survey

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June 1997 aerial photograph of site and surrounding area (reprinted with permission of the Brio Site Task Force).

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**By Glenn F. Huff, Christopher L. Braun, and Roger W. Lee**

**U.S. GEOLOGICAL SURVEY  
Water-Resources Investigations Report 00-4121**

**In cooperation with the U.S. Environmental Protection Agency**

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Charles G. Groat, Director

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**For additional information write to**

**District Chief  
U.S. Geological Survey  
8027 Exchange Dr.  
Austin, TX 78754-4733  
E-mail: [dc\\_tx@usgs.gov](mailto:dc_tx@usgs.gov)**

**Copies of this report can be purchased from**

**U.S. Geological Survey  
Branch of Information Services  
Box 25286  
Denver, CO 80225-0286  
E-mail: [infoservices@usgs.gov](mailto:infoservices@usgs.gov)**

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## VERTICAL DATUM

**Sea level:** In this report, “sea level” refers to the National Geodetic Vertical Datum of 1929—a geodetic datum derived from a general adjustment of the first-order level nets of both the United States and Canada, formerly called Sea Level Datum of 1929.

# Assessment of Potential for Natural Attenuation of Chlorinated Ethenes and Ethanes in Ground Water at a Petrochemical Reclamation Site, Harris County, Texas

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## Abstract

Redox conditions in the Numerous Sand Channels Zone beneath a petrochemical reclamation site in Harris County, Texas, range from sulfate reducing to methanogenic as indicated by the presence of methane in ground water and the range of molecular hydrogen concentrations. Assessment of the potential for reductive dechlorination using BIOCHLOR as a screening tool indicated conditions favoring anaerobic degradation of chlorinated organic compounds in the Numerous Sand Channels Zone. Evidence supporting reductive dechlorination includes apparently biogenic *cis*-1,2-dichloroethene; an increased ratio of 1,2-dichloroethane to 1,1,2-trichloroethane downgradient from the assumed contaminant source area; ethene and methane concentrations greater than background concentrations within the area of the contaminant plume; and a positive correlation of the ratio of ethene to vinyl chloride as a function of methane concentrations. The body of evidence presented in this report argues for hydrogenolysis of trichloroethene to *cis*-1,2-dichloroethene; of 1,1,2-trichloroethane to 1,2-dichloroethane; and of vinyl chloride to ethene within the Numerous Sand Channels Zone.

Simulations using BIOCHLOR yielded apparent first-order decay constants for reductive dechlorination in the sequence

tetrachloroethene→trichloroethene  
→*cis*-1,2-dichloroethene→vinyl chloride→ethene

within the range of literature values reported for each compound and apparent first-order decay constants for reductive dechlorination in the sequence

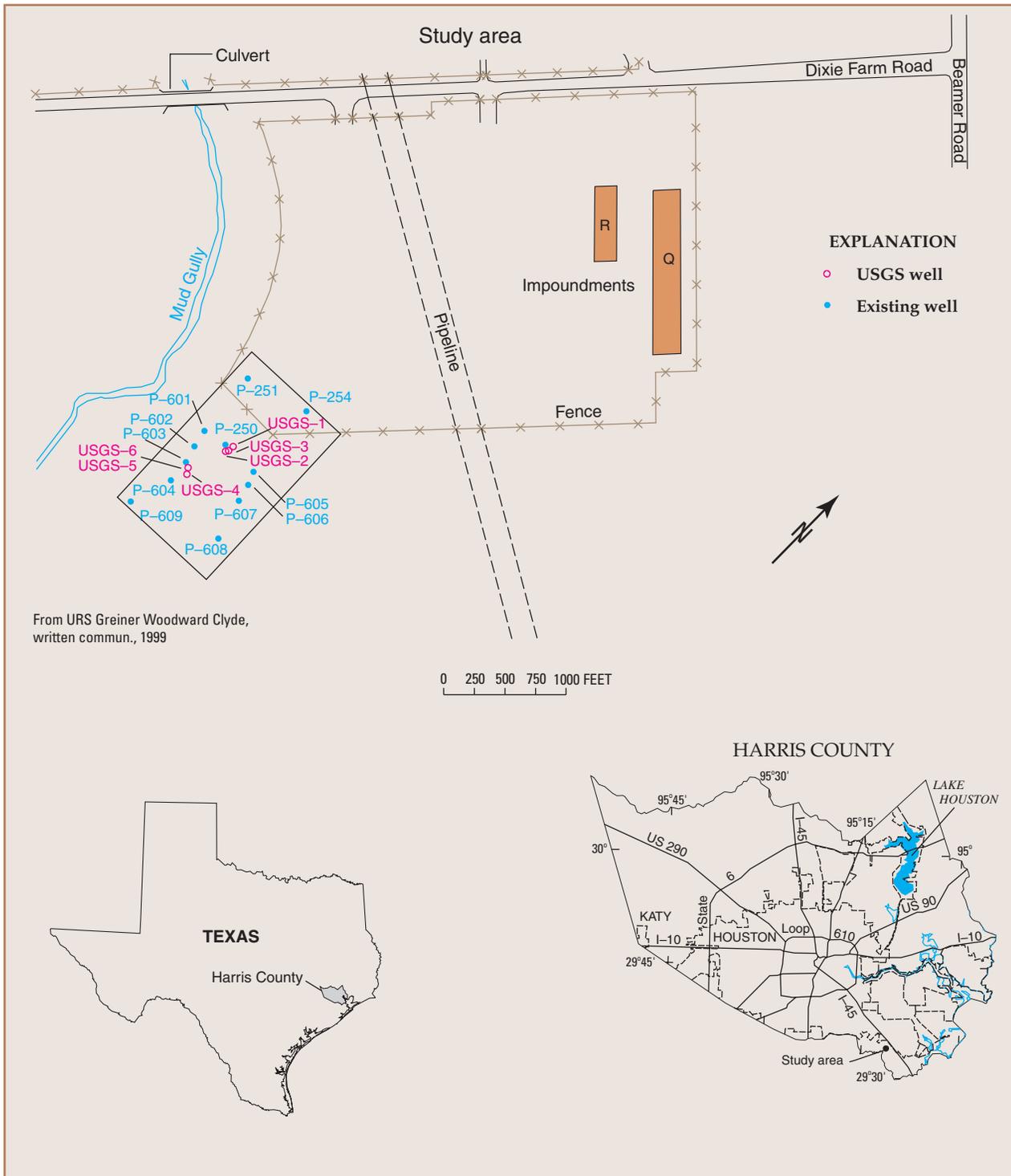
1,1,2-trichloroethane→1,2-dichloroethane

slightly greater than literature values reported for each compound along the upgradient segment of a simulated ground-water flowpath. Except for vinyl chloride, apparent rates of reductive dechlorination for all simulated species show a marked decrease along the downgradient segment of the simulated ground-water flowpath.

Evidence for reductive dechlorination of chlorinated ethenes within the Numerous Sand Channels Zone indicates potential for natural attenuation of chlorinated ethenes. Reductive dechlorination of chlorinated ethanes apparently occurs to a lesser extent, indicating relatively less potential for natural attenuation of chlorinated ethanes. Additional data are needed on the concentrations and distribution of chlorinated ethenes and ethanes in individual fine sand intervals of the Numerous Sand Channels Zone. This information, combined with lower minimum reporting levels for future chloroethane analyses, might enable a more complete and quantitative assessment of the potential for natural attenuation at the site.

## INTRODUCTION

A petrochemical reclamation site (fig. 1, study area) was operated on about 58 acres in Harris County, Texas, from 1957 until 1982. Operations at the study area consisted of reclamation of petrochemical wastes including residues, tank bottoms, and tars generated by off-site refining processes. Many of these wastes were stored in unlined impoundments during the normal



**Figure 1.** Location of the study area and wells.

course of operations at the reclamation site (IT Corporation, 1987).

Ground water in the Numerous Sand Channels Zone (NSCZ), one of five lithologically distinct units beneath the study area, is contaminated with chlorinated ethenes and ethanes. During 1999, the U.S. Geological Survey (USGS), in cooperation with the U.S. Environmental Protection Agency (EPA), collected water-quality samples from and measured water-level altitudes in selected wells to determine the potential for natural attenuation of chlorinated ethenes and ethanes in ground water.

The U.S. Environmental Protection Agency (1999) has defined monitored natural attenuation as, "The reliance on natural attenuation processes (within the context of a carefully controlled and monitored site cleanup approach) to achieve site-specific remediation objectives within a time frame that is reasonable compared to that offered by other more active methods. The natural attenuation processes that are at work in such a remediation approach include a variety of physical, chemical, or biological processes that, under favorable conditions, act without human intervention to reduce the mass, toxicity, mobility, volume, or concentration of contaminants in soil or ground water." A subgrade barrier wall and multiple-layer cap remedy was selected and approved by EPA (John Meyer, U.S. Environmental Protection Agency, written commun., 1999) for remediation of chlorinated ethenes and ethanes contaminating ground water at the petrochemical reclamation site. Some contaminated ground water will remain outside the barrier wall. An assessment of the potential for natural attenuation of chlorinated ethenes and ethanes in ground water outside the barrier wall will aid in determining the long-term effectiveness of this remediation strategy.

## **Purpose and Scope**

This report describes and discusses evidence for the occurrence and rates of natural attenuation of chlorinated ethenes and ethanes within the NSCZ at a petrochemical reclamation site. Occurrence and rates of natural attenuation will be used to assess the potential for natural attenuation as a remedy for ground-water contamination in the study area. The study described herein addresses the behavior of chlorinated ethenes and ethanes within the NSCZ on the basis of ground-water chemical analyses and ground-water-altitude data

collected in early 1999 and data in previously published reports on the study area.

## **Hydrogeology**

The upper 200 feet (ft) of sediments beneath the study area has been subdivided into five lithologically distinct units. These units are (from deepest to shallowest) the Lower Clay Unit, Fifty-foot Sand Zone, Middle Clay Unit, NSCZ, and Upper Clay Unit. Each unit is present over the entire study area (IT Corporation, 1987).

The NSCZ is the shallowest water-yielding zone beneath the study area and contains lower, middle, and upper fine sand intervals separated by clay intervals. The NSCZ ranges from 10 to 20 ft thick over the study area (IT Corporation, 1987). The Upper and Middle Clay Units have relatively low hydraulic conductivities and are thought to form effective confining layers above and below the NSCZ. An estimated 0.01 foot per year (ft/yr) of recharge passes through the Upper Clay Unit into the NSCZ in the study area (IT Corporation, 1987). Aquifer-test data on the fine sand intervals of the NSCZ indicate hydraulic conductivities ranging from 0.80 to 5.2 feet per day (ft/d) (IT Corporation, 1987). Laboratory-determined porosities of sediment samples from the fine sand intervals of the NSCZ, ranging from 37.8 to 46.2 percent (Core Laboratories, 1999), are greater than the range of 10 to 30 percent typical for fine sand (Domenico and Schwartz, 1990). The mass fraction of organic carbon in sediment samples from the fine sand intervals of the NSCZ ranges from 0.13 to 0.53 percent (Core Laboratories, 1999). Bulk densities of sediment samples from the fine sand intervals of the NSCZ range from 2.20 to 2.37 grams per cubic centimeter ( $\text{g}/\text{cm}^3$ ) (Core Laboratories, 1999). The lateral hydraulic gradient within the fine sand intervals of the NSCZ averages about 0.006 over the study area with slightly higher gradients of about 0.01 near Mud Gully (IT Corporation, 1987). Lateral ground-water flow within the NSCZ on the study area is typically either toward or parallel to Mud Gully with a seepage velocity estimated to be about 50 ft/yr (IT Corporation, 1987). The potential for vertical ground-water flow is upward from the Fifty-Foot Sand Zone to the NSCZ over most of the study area (IT Corporation, 1987).

Ground water within the NSCZ downgradient of impoundments Q and R (fig. 1) is contaminated with chlorinated ethenes and ethanes that include

1,1-dichloroethene (1,1-DCE); *trans*-1,2-dichloroethene (*trans*-1,2-DCE); 1,1-dichloroethane (1,1-DCA); 1,2-dichloroethane (1,2-DCA); trichloroethene (TCE); and 1,1,2-trichloroethane (1,1,2-TCA) (IT Corporation, 1987). These contaminants are thought to be leached from material stored in the impoundments within the study area and infiltrated into the ground water. Chlorinated ethenes and ethanes found in association with impoundment materials include vinyl chloride (VC); 1,1-DCE; *trans*-1,2-DCE; 1,1-DCA; 1,2-DCA; TCE; 1,1,2-TCA; and tetrachloroethene (PCE) (IT Corporation, 1987).

## Reductive Dechlorination

Substantial research within the last two decades has increased the understanding of processes governing the fate of halogenated aliphatic hydrocarbons in ground water. Discussion of these processes is presented by Vogel and others (1987), Sims and others (1991), Chapelle (1993), and Wiedemeier and others (1999). Numerous laboratory investigations (Vogel and McCarty, 1985; Freedman and Gossett, 1989; Bagley and Gossett, 1990; DeBrunin and others, 1992; Bradley and Chapelle, 1997) and field investigations (Cox and others, 1995; Lee and others, 1995; Weaver and others, 1996; Imbrigiotta and others, 1996) have documented that chlorinated ethenes can undergo reductive dechlorination reactions under anaerobic conditions. Anaerobic reductive dechlorination reactions involving chlorinated ethanes have also been documented in laboratory studies (Belay and Daniels, 1987; Schanke and Wackett, 1992; Tandol and others, 1994; Chen and others, 1996; Lorah and Olsen, 1999) and field studies (Lorah and Olsen, 1999).

Reductive dechlorination of chlorinated ethanes generally takes place either by direct substitution of hydrogen for chlorine (hydrogenolysis), as exemplified by the transformation of 1,1,1,2-tetrachloroethane (PCA) to 1,1,2-TCA, or by elimination of adjacent chlorines and formation of an ethene (dichloroelimination), as exemplified by the transformation of 1,1,2-TCA to VC. Reductive dechlorination of chlorinated ethenes generally takes place by hydrogenolysis as exemplified by the transformation of VC to ethene (Vogel and others, 1987; Chen and others, 1996; Lorah and Olsen, 1999). Anaerobic conditions (measured dissolved oxygen [DO] less than 0.5 milligram per liter [mg/L]) and the presence of molecular hydrogen generated through microbially mediated fermentation reactions are posi-

tive indicators of conditions favoring reductive dechlorination. Sequential dechlorination of a polychlorinated ethene or ethane generally requires a progressively more reducing environment. For example, a more strongly reducing environment is required to produce VC from 1,2-dichloroethene (1,2-DCE) than is required to produce 1,2-DCE from TCE. Selected examples from the literature illustrating this generalization include (1) reductive dechlorination of PCE and TCE under conditions as mild as nitrate reducing (Vogel and others, 1987), and (2) reductive dechlorination of *cis*-1,2-dichloroethene (*cis*-1,2-DCE) to VC and VC to ethene in more strongly reducing conditions ranging from iron III reducing to methanogenic (Freedman and Gossett, 1989; DiStefano and others, 1991; DeBrunin and others, 1992; Bradley and Chapelle, 1996).

Many sites having ground water contaminated with chlorinated alkenes or alkanes have a history of contamination spanning many years. Therefore, if conditions for anaerobic microbial growth are favorable, microbial populations attendant on reductive dechlorination could be in place. The presence of molecular hydrogen as an electron donor and the reduction-oxidation (redox) state of the environment are the variables that govern the potential for reductive dechlorination of chlorinated alkenes and alkanes.

Historically, platinum electrode measurements have been the most widely used technique for determining the redox state of aqueous systems. Measurements made using platinum electrodes can be used to distinguish between generally aerobic and anaerobic conditions but cannot adequately distinguish between various reducing redox states such as iron III reducing, sulfate reducing, or methanogenic. Microbially mediated fermentation reactions in anaerobic ground-water systems produce concentrations of molecular hydrogen that are characteristic of differing redox states. The following method described by Chapelle and others (1995) and Chapelle (1996) uses trends in dissolved concentrations of redox reactive constituents including DO, nitrate, iron, sulfate, and sulfide along with concentrations of methane and molecular hydrogen to determine the redox state of a ground-water system. If measured DO concentrations in ground water are greater than 0.5 mg/L, reductive dechlorination is an unlikely process. Nitrate concentrations greater than 0.5 mg/L and molecular hydrogen concentrations less than 0.1 nanomole per liter (nM/L) indicate nitrate-reducing conditions. An increase in the concentrations of iron II along a ground-water flowpath (in the absence of iron II sinks

such as sulfide), coupled with molecular hydrogen concentrations in the 0.2 to 0.6 nM/L range, indicate iron III-reducing conditions. Decreasing concentrations of sulfate (in the absence of a sulfate source such as gypsum) or increasing concentrations of sulfide along a ground-water flowpath, coupled with molecular hydrogen concentrations in the 1 to 4 nM/L range, indicate sulfate-reducing conditions. Finally, the presence of methane along a ground-water flowpath, coupled with molecular hydrogen concentrations greater than 5 nM/L, indicates methanogenic conditions.

### Sampling and Analytical Methods

To determine the redox environments in ground water that could affect natural attenuation, the data listed in table 1 (at end of report) were determined in the field or in the laboratory, as indicated. Ground-water samples were collected from wells using a peristaltic pump. Wells were pumped for a minimum of 30 minutes at 0.5 liter per minute (L/min) prior to sampling.

Decontamination procedures consisted of rinsing equipment between wells with deionized water and a dilute detergent solution (when needed). Decontaminated equipment included glass bulbs for headspace sample collection, monitoring probes, and glassware used with Hach colorimeters. Wells were equipped with dedicated Teflon tubing, and new Tygon tubing was used to attach to the peristaltic pump for each well sampled.

### Dissolved Oxygen

Concentrations of DO were measured using the CHEMetrics indigo carmine colorimetric method (Gilbert and others, 1982). While ground water was pumped into an overflowing sampling cell, an ampule was placed tip down into the sampling cell. The tip of the ampule was broken to allow a fixed amount of sample to be drawn into the ampule. The broken tip was kept submerged to avoid contamination with atmospheric oxygen. After the ampule was filled, the tip was covered, and the ampule was inverted several times to mix the ground-water sample with the color-developing agents pre-loaded into the ampule. After allowing the sample and color-developing agents to react for 2 minutes, the ampule was placed in the CHEMetrics VRR multivariate photometer for analysis, and the resulting DO concentration was recorded in the field notebook. The lower detection limit for DO using this method was 0.05 mg/L.

### Total Iron Species and Iron II

Concentrations of total iron species and total iron II were measured using the Hach FerroVer colorimetric method for total iron species and the Hach phenanthroline powder pillow method for total iron II (Hach, 1989). In these methods, 25 milliliters (mL) of water sample is added to four colorimetric vials. One vial was used to determine total iron species, one vial was used to determine total iron II, and the two remaining vials were used as instrument blanks. Concentrations of total iron species were determined by first adding the total iron reagent to a vial and allowing color to develop for 3 minutes. During color development, the instrument blank vial was placed in the DR/2000 spectrophotometer, and zero concentration was set at the 510-nanometer (nm) wavelength. After color development was complete, the blank vial was removed, and the total iron vial was placed into the spectrophotometer. The resulting concentrations of total iron species were recorded in the field notebook. If the determined concentration of total iron species was greater than or equal to 0.01 mg/L, total iron II concentrations were determined by the same procedure as for total iron species, but using iron II reagent rather than total iron reagent. The upper method detection limit of total iron species and total iron II is 3 mg/L. If a sample concentration was greater than the upper method detection limit, a 50-percent dilution of the sample with deionized water was used, and the sample was re-analyzed.

### Total Sulfide

The total sulfide concentration was measured using the CHEMetrics methylene blue colorimetric method (U.S. Environmental Protection Agency, 1983). Three drops of the sulfide reagent were added to a sample cell containing 25 mL of ground water and mixed by stirring with the tip of the ampule containing the sulfide reagent. After mixing, the tip of the ampule was snapped against the side of the sample cell, and the ampule was filled. The ampule was inverted several times to mix the reagent and sample. The methylene blue color was allowed to develop for 5 minutes. After 5 minutes, a blank sample was placed into the VRR multivariate photometer. The blank sample was removed, the sample cell containing the ground-water sample was placed in the colorimeter, and the concentration of total sulfide was read and recorded in the field notebook. The lower and upper detection limits for total

sulfide using the method and instrumentation listed above are 0.001 and 1.5 mg/L, respectively.

### Molecular Hydrogen

The concentration of molecular hydrogen was measured using the “bubble strip” method (Chapelle, 1996). This method entails pumping ground water through a glass water-sampling bulb fitted with a septum. Once flow was established at about 500 milliliters per minute (mL/min) and the bulb was filled and free of gas bubbles, about 20 mL of hydrogen-free helium was injected through the septum. The induced headspace was allowed to equilibrate with the flowing ground water for about 20 minutes. After equilibration, about 10 mL of the headspace gas was withdrawn using a 10-mL gas-tight syringe that closed with a valve. After 5 minutes, a second 10-mL sample of headspace gas was withdrawn into a second gas-tight syringe. Both samples were analyzed for molecular hydrogen by injection of about 6 mL of sampled headspace gas into a RGA3 reducing-gas-detector chromatograph that was calibrated using standard gases.

### Methane and Carbon Dioxide

Concentrations of methane and carbon dioxide were measured by injecting the remainder of the 10-mL headspace gas sample collected from the glass bulb for molecular hydrogen analysis into an MTI dual-column gas chromatograph equipped with a thermal conductivity detector. Using the procedure described above, concentrations of methane and carbon dioxide can be detected at levels as low as 0.1 micromole per liter ( $\mu\text{M/L}$ ).

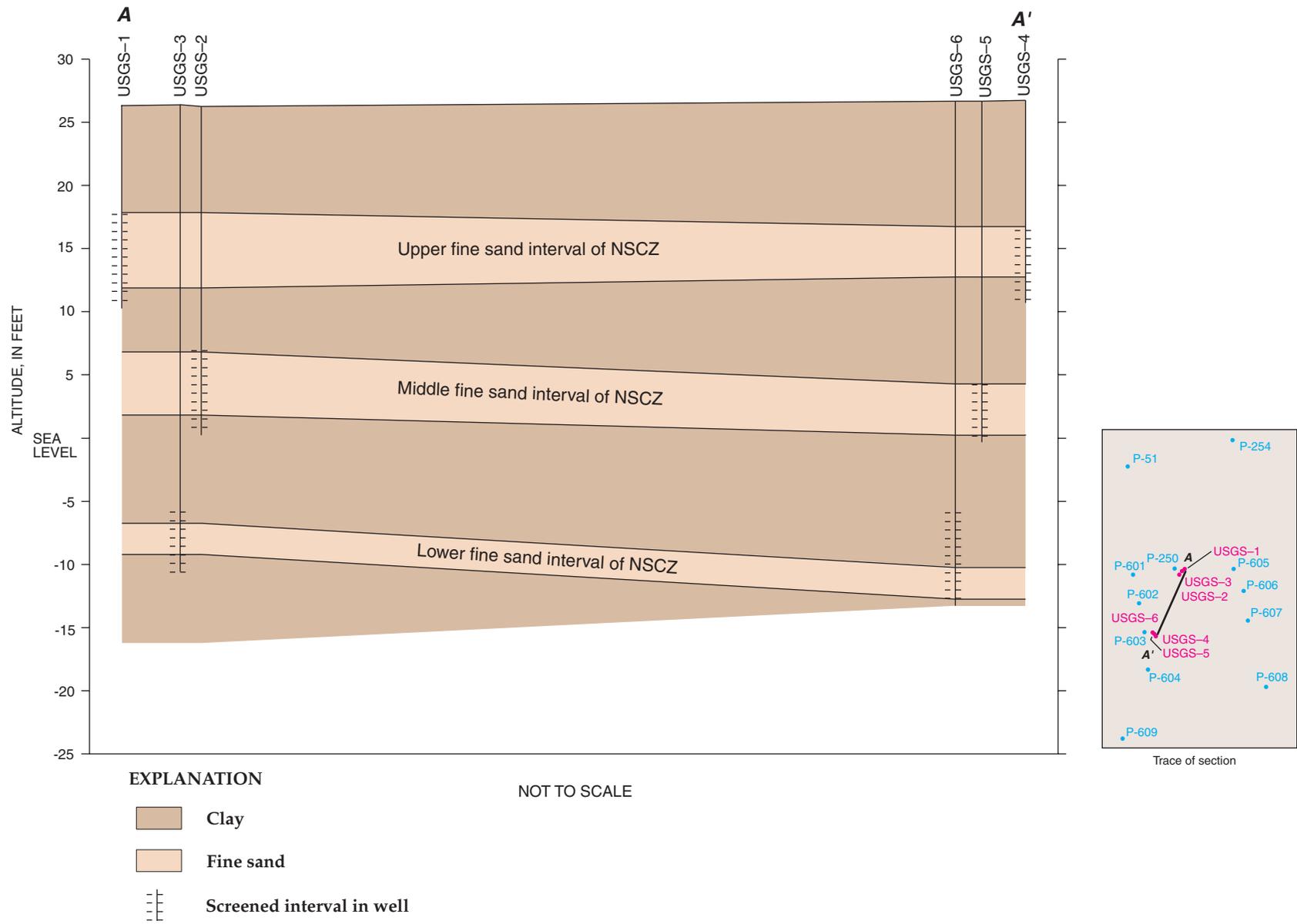
## POTENTIAL FOR NATURAL ATTENUATION OF CHLORINATED ETHENES AND ETHANES

Ground-water altitudes measured on March 17, 1999, are listed in table 2 (at end of report). The lithologic section A–A' on figure 2 shows wells USGS–1, USGS–2, USGS–3, USGS–4, USGS–5, and USGS–6 screened in the NSCZ. Each cluster of three wells contains one well that is screened in only one of the three fine sand intervals in the NSCZ. Some of the remaining wells at the study area have screened intervals as much as 25 ft long (table 2) and are typically open to more than one fine sand interval within the NSCZ. The locations of selected wells sampled during February 8–10,

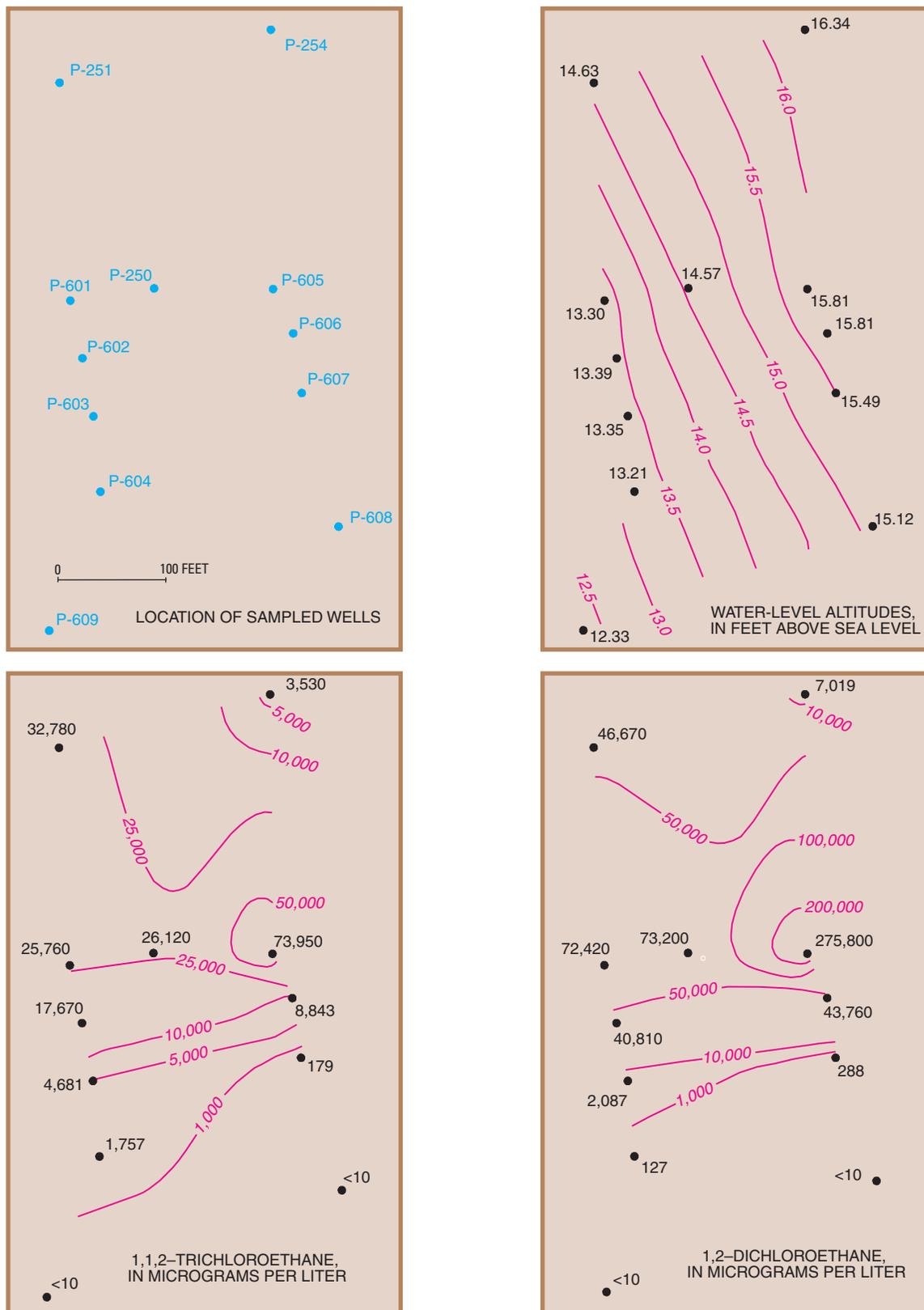
1999, are shown in figure 3. Results of sampling for volatile organic compounds, field properties and constituents, inorganic ions, and dissolved organic carbon for all wells sampled are listed in tables 3 and 4 (at end of report). All chemical analyses shown in table 3 and chemical analyses of dissolved chloride, dissolved sulfate, dissolved nitrite plus nitrate, and dissolved organic carbon shown in table 4 were performed by an EPA-approved contract laboratory. Specific methods used in these chemical analyses include EPA method 8260 (volatile organic compounds), EPA method 8015 (ethene, ethane, and 2-chloroethanol), EPA method 415.1 (dissolved organic carbon), EPA method 325.3 (dissolved chloride), EPA method 353.2 (dissolved nitrite plus nitrate, as N), and EPA method 375.4 (dissolved sulfate). All other chemical analyses were performed by the U.S. Geological Survey.

### Spatial Distribution

Water-level-altitude contours and lines of equal concentrations of selected chlorinated ethenes and ethanes in ground water from the sampled wells are shown in figure 3. Concentrations of chlorinated ethenes and ethanes from the USGS wells were not used to generate the lines of equal concentrations in figure 3 because the USGS wells are screened in discreet intervals of the NSCZ, whereas the non-USGS wells have fully penetrating screens. TCE; *trans*-1,2-DCE; *cis*-1,2-DCE; 1,1-DCE; and VC are the chlorinated ethenes with the highest concentrations in wells shown in figure 3. The highest concentrations of chlorinated ethanes are for 1,1,2-TCA and 1,2-DCA. The highest concentrations of all compounds shown in figure 3 are generally in the sample from well P–605. The approximate axis of the contaminant plume is aligned roughly parallel to the local direction of ground-water flow as inferred from contours of ground-water altitudes and is approximated by the ground-water flowpath from well P–605, past well P–250, to well P–601. The concentrations of 1,1,2-TCA; 1,2-DCA; 1,1-DCA; TCE; *trans*-1,2-DCE; *cis*-1,2-DCE; 1,1-DCE; VC; and PCE in wells along the approximate ground-water flowpath are shown in figure 4. The contaminant plume likely exists both upgradient and downgradient of the area of the sampled wells. Apparent transverse spreading of all chlorinated ethenes and ethanes shown in figure 3 is generally perpendicular to the local direction of ground-water flow. The apparent transverse extent of the contaminant plume varies for each organic compound.



**Figure 2.** Lithologic section A–A' showing wells screened in the Numerous Sand Channels Zone (NSCZ).



**Figure 3.** Location of sampled wells, water-level-altitude contours, and lines of equal concentrations of selected chlorinated ethenes and ethanes in samples from the wells.

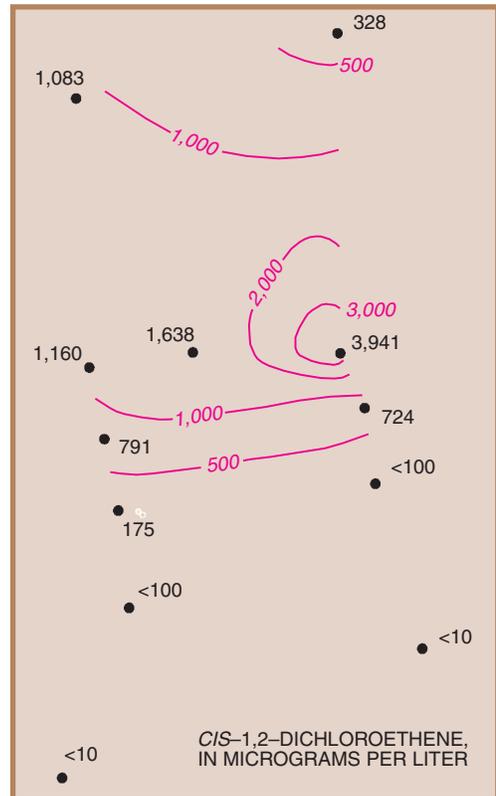
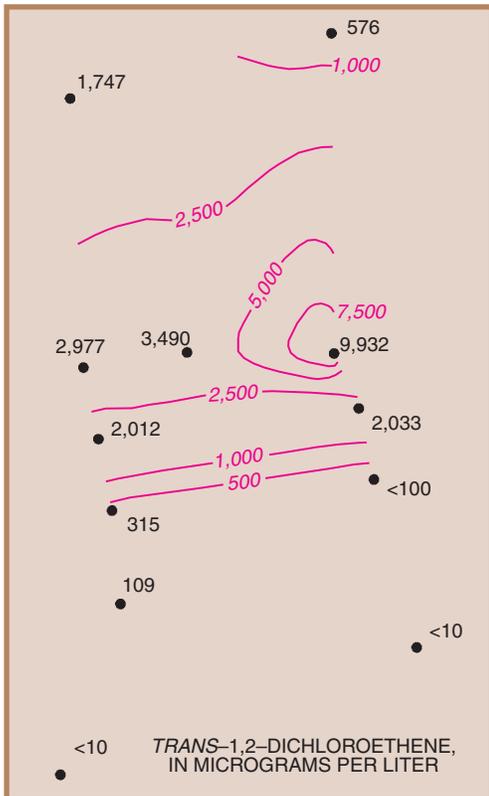
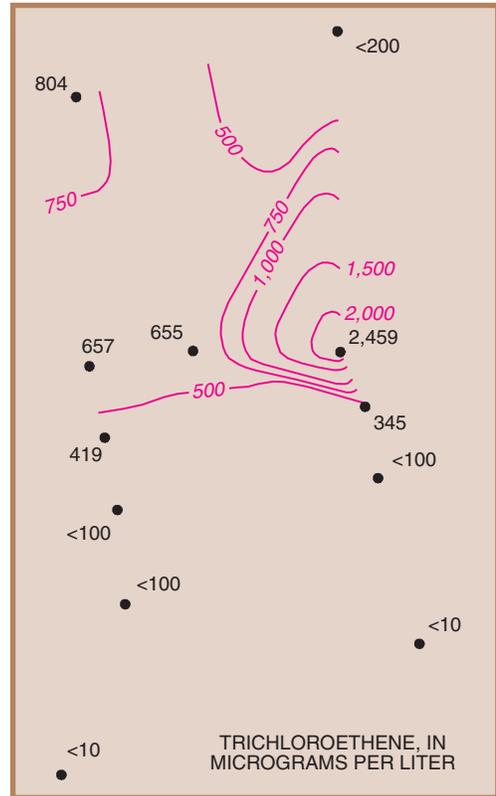
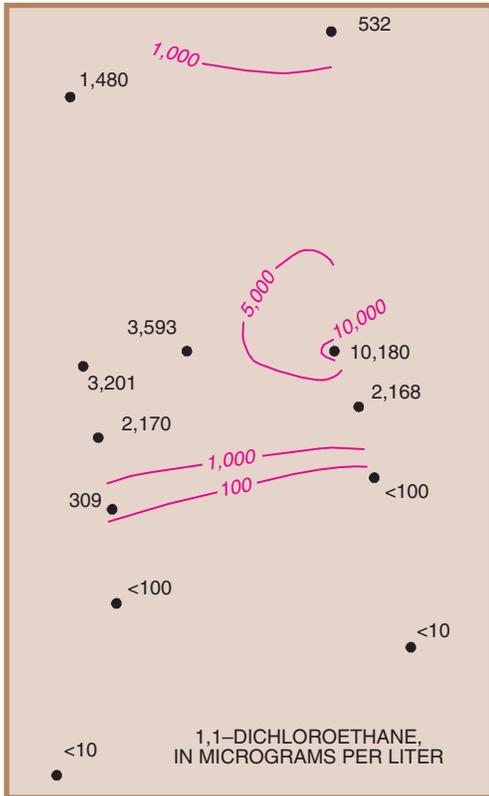


Figure 3.—Continued.

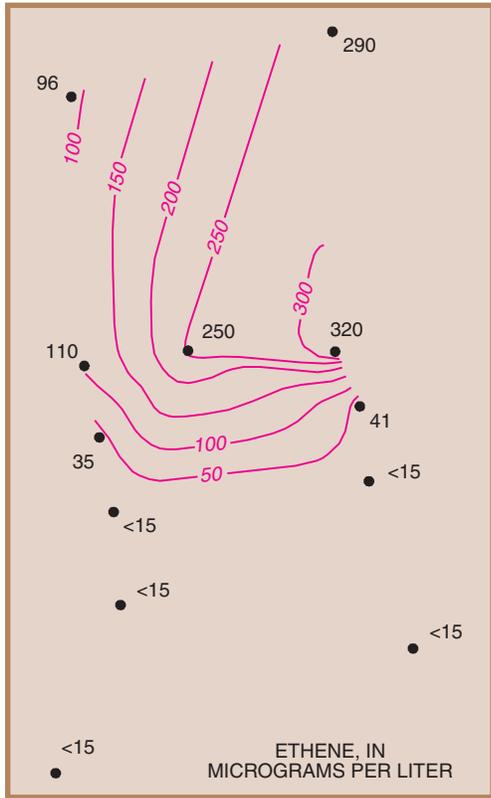
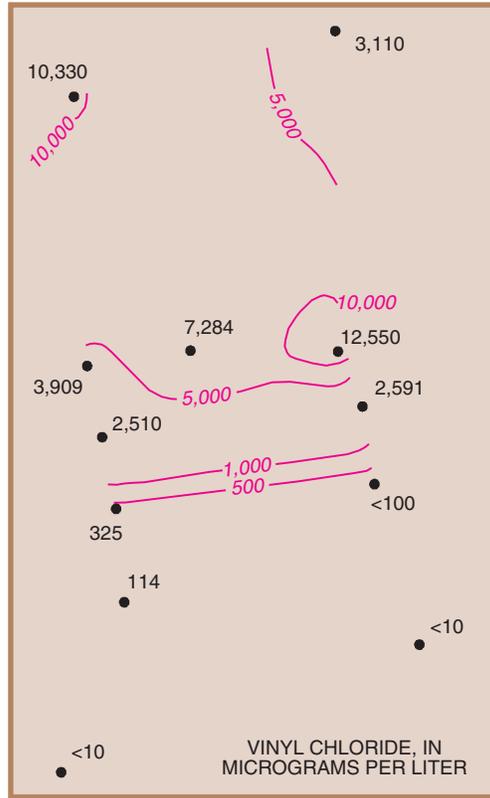
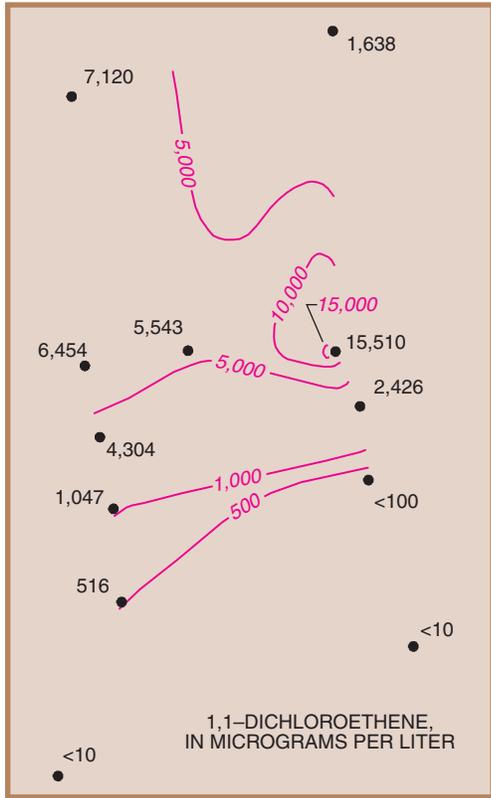
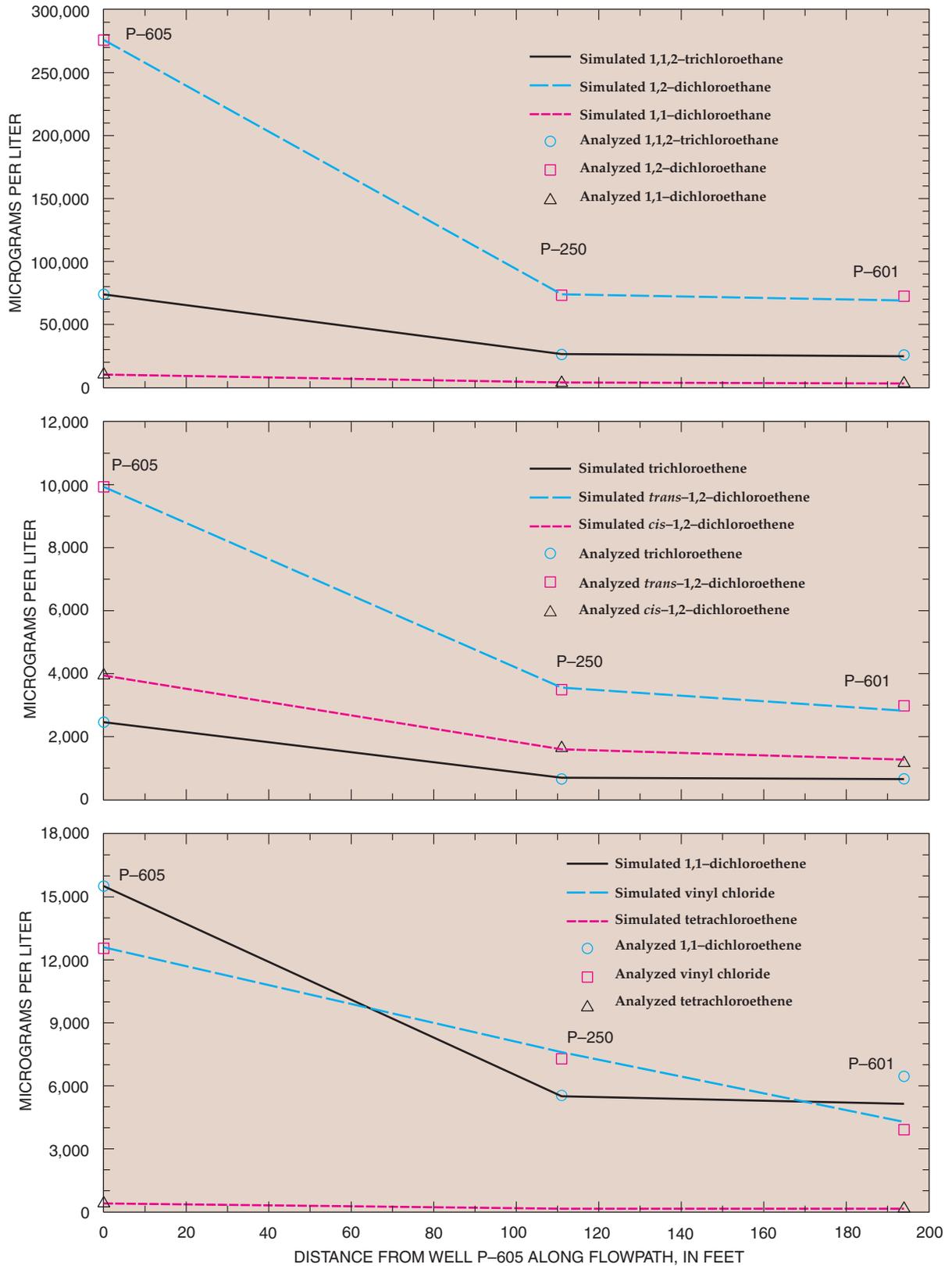


Figure 3.—Continued.



**Figure 4.** Simulated and analyzed concentrations of selected chlorinated ethenes and ethanes along the groundwater flowpath.

Concentrations of chlorinated ethenes and ethanes are generally higher in the lower fine sand interval (well USGS-3) than in the middle fine sand interval (well USGS-2) of the NSCZ near the plume axis and higher in the middle fine sand interval (well USGS-5) than in the lower fine sand interval (well USGS-6) of the NSCZ nearer the plume margin. This variation in the transverse distribution of contaminants could indicate a higher degree of transverse dispersivity in the middle fine sand interval of the NSCZ or could reflect some other source of inhomogeneity in the internal structure of the plume. Because no water could be produced from wells USGS-1 and USGS-4 during the February sampling, no information is available on the concentrations of chlorinated ethenes and ethanes in the upper fine sand interval of the NSCZ.

### Redox Conditions

Concentrations of DO in all sampled wells were less than or equal to 0.37 mg/L (table 4) indicating anaerobic conditions within the NSCZ. Efforts at collection of field properties and constituents in the wells along the approximate ground-water flowpath yielded: inadequate sample volume for determination of field properties and constituents from well P-250; samples containing concentrations less than 0.1 mg/L total iron species in wells P-601 and P-605; and samples containing concentrations of total sulfide of 0.07 mg/L and less than the method detection limit in wells P-601 and P-605, respectively. All ground-water samples from the NSCZ contained dissolved nitrite plus dissolved nitrate concentrations less than the method detection limit of 0.1 mg/L. All non-USGS wells sampled for methane contained detectable concentrations except for wells P-608 and P-609, both of which are beyond the apparent edge of the contaminant plume. Concentrations of molecular hydrogen in samples from non-USGS wells ranged from 0.06 to 4,765 nM/L. Many of the non-USGS wells yielded water with molecular hydrogen concentrations much higher than can be explained by microbial activity. Construction records show that non-USGS wells sampled for this study were cased using galvanized iron pipe and were equipped with stainless steel screens, whereas USGS wells were constructed entirely of polyvinyl chloride (PVC). It is therefore reasonable to conclude that molecular hydrogen concentrations greater than can be explained by microbial activity originated from corrosion of the iron well casings and possibly the steel screens. As a result, molecular hydro-

gen concentrations of samples collected from non-USGS wells could not be used in defining the redox state of the NSCZ. Molecular hydrogen concentrations in samples from wells USGS-2 and USGS-5 indicated a sulfate-reducing redox state in the middle fine sand interval of the NSCZ. Molecular hydrogen concentrations in samples from wells USGS-3 and USGS-6 indicated a methanogenic redox state in the lower fine sand interval of the NSCZ. The wide-spread detection of methane in samples from wells and the molecular hydrogen data from the USGS wells indicated the redox environment within the NSCZ is sulfate-reducing to methanogenic.

### One-Dimensional Fate and Transport Modeling

For a one-dimensional system with flow in the  $x$  direction, the concentration of a solute undergoing dispersion, advection, retardation by sorption, and degradation while moving through a porous medium is given by the solution to the transport equation

$$D \frac{\partial^2 C}{\partial x^2} - v \frac{\partial C}{\partial x} = R \left( \frac{\partial C}{\partial t} + \lambda C \right) \pm W, \quad (1)$$

where

$D$  is the dispersion coefficient,

$C$  is the solute concentration,

$v$  is the seepage velocity,

$R$  is the retardation coefficient,

$t$  is the time,

$\lambda$  is the first-order decay constant, and

$W$  is a source/sink term. (All components of equation should be expressed in consistent units.)

Analytical solutions to equations similar to the one above have been presented for the study of transport of reactive contaminants in ground water by Domenico (1987), Buscheck and Alcantar (1995), Weaver and others (1996), and Sun and others (1999).

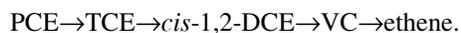
BIOCHLOR (Aziz and others, 1999) is a screening model that simulates the degradation of chlorinated solvents. The software is programmed to use the Domenico (1987) analytical solute transport model with sequential first-order decay for simulating in-place biodegradation (Sun and others, 1999). Data required for the model include hydraulic conductivity, hydraulic gradient, effective porosity, modeled area length and width, soil bulk density, fraction organic carbon, and

upgradient and downgradient concentrations of chlorinated solvents.

Assessment of the potential for reductive dechlorination of chlorinated ethenes and ethanes at the study area using BIOCHLOR as a screening tool yielded a score of 21, indicating “strong evidence for anaerobic degradation of chlorinated organic compounds” (Aziz and others, 1999). Provided the source of contaminants found in wells sampled for this study is similar to those found in impoundments Q and R, evidence supporting reductive dechlorination of chlorinated ethenes and ethanes includes the reported lack of *cis*-1,2-DCE in soil from impoundments Q and R (IT Corporation, 1987) contrasted with the detection of *cis*-1,2-DCE in samples from wells P-605, P-250, and P-601. Also, the higher concentration of 1,1,2-TCA (65,700 milligrams per kilogram [mg/kg]) compared to 1,2-DCA (8,140 mg/kg) in soil from impoundment Q (IT Corporation, 1987) contrasts with the higher concentrations of 1,2-DCA compared to 1,1,2-TCA in samples from wells P-605, P-250, and P-601. Additional evidence supporting reductive dechlorination includes ethene and methane concentrations greater than background concentrations in water samples from within the area of the contaminant plume and a positive correlation of the ratio of ethene to VC as a function of methane concentrations. The body of evidence listed above argues for hydrogenolysis of TCE to *cis*-1,2-DCE; hydrogenolysis of 1,1,2-TCA to 1,2-DCA; and hydrogenolysis of VC to ethene within the NSCZ. The presence of 5,100 mg/kg of 1,2-DCA in soil from impoundment R (IT Corporation, 1987) indicates that 1,2-DCA could also have been released as an initial contaminant.

The behavior of 1,1,2-TCA; 1,2-DCA; 1,1-DCA; TCE; *trans*-1,2-DCE; *cis*-1,2-DCE; 1,1-DCE; VC; and PCE was simulated along the approximate ground-water flowpath of wells P-605 to P-250 to P-601 using BIOCHLOR. BIOCHLOR calculated an R value of 3.4 for all simulated compounds using provided data on sediment bulk density ( $2.27 \text{ g/cm}^3$ ) and percent organic carbon (0.16 percent) (Core Laboratories, 1999) and using data from literature for values of octanol-water partitioning coefficients (Aziz and others, 1999). Longitudinal dispersivity was set to 10 percent of the simulated ground-water flowpath length, and transverse dispersivity was set to 10 percent of the longitudinal dispersivity in each simulation. Contaminant source width was assumed to be 100 ft in each simulation. Seepage velocity was set to 50 ft/yr in each simulation in accordance with reported estimates (IT Corporation, 1987).

Steady-state conditions were assumed for all simulations. The simulation scheme used in BIOCHLOR involves sequential reductive dechlorination of a series of parent and daughter products. For chlorinated ethenes the sequence is



For chlorinated ethanes the sequence is



The reductive dechlorination of TCE was simulated to produce only *cis*-1,2-DCE on the basis of the documented predominant production of the *cis* isomer of 1,2-DCE during biodegradation of TCE (Parsons and Barrio-Lage, 1984; Parsons and others, 1984; Parsons and others, 1985; Barrio-Lage and others, 1986, 1987). The production rate of each daughter compound is calculated from the concentration and decay rate of its parent compound. The reductive dechlorination of 1,1-DCA, *trans*-1,2-DCE, and 1,1-DCE were simulated as unsupported compounds (that is, no parent compounds were present).

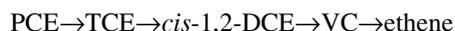
Initial attempts to simulate changes in concentrations of chlorinated ethenes and ethanes along the entire approximate ground-water flowpath using concentrations in well P-605 as the source term in the sequences described above were largely unsuccessful. Simulations using a single value for the apparent first-order decay constant of each compound consistently overpredicted compound concentrations in well P-250 and underpredicted compound concentrations in well P-601. However, simulation of VC concentrations over the entire approximate ground-water flowpath using a single apparent decay constant of 0.39 per year for VC gave reasonable results.

Subsequent simulations divided the approximate ground-water flowpath into an upgradient segment between wells P-605 and P-250 and a downgradient segment between wells P-250 and P-601. Concentrations of chlorinated ethenes and ethanes in samples from well P-605 were used as a source term for simulations of changes in concentrations along the upgradient segment. Concentrations in samples from well P-250 were used as a source term for simulations of changes in concentrations along the downgradient segment. Advantages of simulation along segments include the ability to more closely match simulated and analytical results. Disadvantages include the inability to establish definitively that changes in concentrations along the segments can be represented as partial functions of first-order decay. Results of simulations along the segments

of the approximate ground-water flowpath are shown in figure 4. Apparent first-order decay constants, which gave simulated concentrations in best agreement with observed changes in concentrations along the segments of the approximate ground-water flowpath, are listed in table 5. Apparent first-order decay constants for selected chlorinated ethenes derived from simulation of changes in concentrations along the upgradient section of the approximate ground-water flowpath are within the range of literature values reported for each compound (Dupont and others, 1996; Ellis and others, 1996; Weaver and others, 1996; Wiedemeier and others, 1996). Apparent first-order decay constants for chlorinated ethanes derived from simulation of changes in concentrations along the upgradient section of the approximate ground-water flowpath are slightly greater than literature values reported for each compound (Vogel and McCarty, 1987; McCarty, 1996). Changes in concentrations along the downgradient section of the approximate ground-water flowpath are generally best simulated assuming no reductive dechlorination or much slower reductive dechlorination than along the upgradient section. Simulated apparent first-order decay constants for VC are similar on both segments.

The apparent first-order decay constants for VC were calculated assuming contributions of VC from the decay of *cis*-1,2-DCE, *trans*-1,2-DCE, and 1,1-DCE for the upgradient segment and from the decay of *cis*-1,2-DCE and *trans*-1,2-DCE for the downgradient segment. Apparent analytical concentrations of 1,1-DCE indicated no degradation between wells P-250 and P-601. The concentration of PCE in samples from well P-250 was reported as less than the method detection limit of 200 micrograms per liter ( $\mu\text{g/L}$ ). For simulation purposes, water in well P-250 was assumed to have a PCE concentration of 148  $\mu\text{g/L}$ , identical to that reported for water in well P-601. Any error in the apparent first-order decay constant calculated for TCE caused by this assumption was likely small given the relatively low concentration of PCE. Simulated concentrations of all compounds were relatively insensitive to changes in values of dispersivity and retardation coefficient and relatively sensitive to changes in seepage velocity and apparent first-order decay constant.

All parent and daughter compounds in the sequence



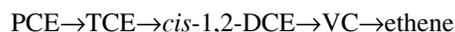
are present in samples from the NSCZ, supporting the plausibility of reductive dechlorination. Documenting reductive dechlorination in the sequence



is more difficult. Neither CA nor ethane were detected in concentrations greater than the method detection limits in any NSCZ sample. The method detection limits were 100 and 200  $\mu\text{g/L}$  for CA samples from within the plume and 16  $\mu\text{g/L}$  for all ethane samples. If reductive dechlorination of 1,2-DCA to CA were taking place, the subsequent breakdown of CA and ethane would need to be extremely rapid. Alternatively, 1,2-DCA can undergo dichloroelimination to ethene (Tandol and others, 1994; Chen and others, 1996). Reductive dechlorination products of 1,1-DCA and 1,2-DCA could not be identified using the data collected in this study. However, it appears that much of the observed 1,2-DCA might be derived from hydrogenolysis of 1,1,2-TCA.

In addition to reductive dechlorination, observed decreases in concentrations of contaminants along the upgradient segment of the approximate ground-water flowpath could be partially explained by inhomogeneity of contaminants within the NSCZ. Concentrations of contaminants in samples from well USGS-2 (middle fine sand interval of the NSCZ) are consistently lower than in samples from well USGS-3 (lower fine sand interval of the NSCZ). Any preferential production of water by non-USGS wells from either the middle or lower fine sand intervals could thus affect apparent analytical concentrations. This could, in turn, affect apparent decay-rate calculations.

Apparent reductive dechlorination in the sequence



indicates the potential for natural attenuation of chlorinated ethenes at the study area. Available data indicate that reductive dechlorination in the sequence



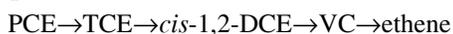
might not proceed beyond the transformation of 1,1,2-TCA to 1,2-DCA. If so, natural attenuation might have relatively less potential in the remediation of chlorinated ethanes. Further sampling and analysis, in which the minimum reporting level for CA from samples within the plume could be decreased from the 100 and 200  $\mu\text{g/L}$  used in this study, might provide a more complete picture of the potential for natural attenuation of chlorinated ethenes at the study area. Additional data on the concentrations of chlorinated ethenes and ethanes within individual fine sand intervals of the NSCZ might

provide a more accurate picture of the distribution of subsurface contaminants. This might, in turn, enable a more accurate and quantitative assessment of the potential for natural attenuation as a remediation strategy.

## CONCLUSIONS

Redox conditions within the NSCZ range from sulfate reducing to methanogenic as indicated by the presence of methane in ground water and range of molecular hydrogen concentrations in water samples from PVC-cased wells. Molecular hydrogen concentrations in water samples from many of the non-PVC cased wells were greater than can be explained by microbial activity. These molecular hydrogen concentrations likely result from corrosion of the well casings or screens, and therefore cannot be used to define the redox state of the NSCZ. The sulfate-reducing to methanogenic redox state of the NSCZ favors reductive dechlorination of chlorinated ethenes and ethanes in ground water. Assessment of the potential for reductive dechlorination of chlorinated ethenes and ethanes within the NSCZ, using BIOCHLOR as a screening tool, indicated conditions favoring anaerobic degradation of chlorinated organic compounds. Evidence supporting reductive dechlorination includes apparently biogenic *cis*-1,2-DCE; an increased ratio of 1,2-DCA to 1,1,2-TCA downgradient from the assumed contaminant source area; ethene and methane concentrations greater than background concentrations within the area of the contaminant plume; and a positive correlation of the ratio of ethene to VC as a function of methane concentrations. The body of evidence presented in this report argues for hydrogenolysis of TCE to *cis*-1,2-DCE; hydrogenolysis of 1,1,2-TCA to 1,2-DCA; and hydrogenolysis of VC to ethene within the NSCZ.

Simulations using BIOCHLOR yielded apparent first-order decay constants for reductive dechlorination in the sequence



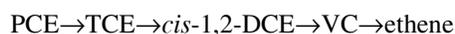
within the range of literature values reported for each compound and apparent first-order decay constants for reductive dechlorination in the sequence



slightly greater than literature values reported for each compound over the upgradient segment of the approximate ground-water flowpath. Except for VC, apparent rates of reductive dechlorination of ethenes and ethanes are more rapid over the upgradient segment (wells P-605 to P-250) than over the downgradient segment

(wells P-250 to P-601) of the approximate ground-water flowpath. Reductive dechlorination rates of VC are similar over both approximate ground-water flowpath segments.

Apparent reductive dechlorination in the sequence



indicates the potential for natural attenuation of chlorinated ethenes. Available data indicate that reductive dechlorination in the sequence



might not proceed beyond the transformation of 1,1,2-TCA to 1,2-DCA. If so, natural attenuation might have relatively less potential in the remediation of chlorinated ethanes. Further sampling and analysis, in which the minimum reporting level for CA from samples within the plume could be decreased from the 100 and 200  $\mu\text{g/L}$  levels used in this study, might provide a more complete picture of the potential for natural attenuation of chlorinated ethanes.

There is evidence for lateral and vertical inhomogeneity in the distribution of chlorinated ethenes and ethanes within the NSCZ. These inhomogeneities could affect apparent decay-rate calculations. Additional data on the concentrations and distributions of chlorinated ethenes and ethanes within individual fine sand intervals of the NSCZ might enable a more accurate picture of the distribution of subsurface contaminants at the study area. This additional information might enable a more accurate and quantitative assessment of the potential for natural attenuation.

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**Table 1.** Properties and constituents measured to evaluate the potential for natural attenuation of chlorinated ethenes and ethanes in ground water

Property or constituent	Method of analysis
Dissolved oxygen	CHEMetrics indigo carmine colorimetric method
Ferrous iron and total iron	Hach colorimetric methods 255 and 265
Sulfide	CHEMetrics methylene blue colorimetric method
Hydrogen	Field gas chromatography, reduction gas detection
Methane, carbon dioxide	Field gas chromatography, thermal conductivity detection
Volatile organic compounds	Gas chromatography/mass spectroscopy laboratory analysis, method 8260
Ethene	Gas chromatography/mass spectroscopy laboratory analysis, method 8015

**Table 2.** Selected data for wells

[--, no data; negative data indicate feet below sea level]

Well no. or identifier (fig. 1)	Altitude of land surface (feet above sea level)	Altitude of top of well casing (feet above sea level)	Altitude of top of screened interval (feet above sea level)	Length of screened interval (feet)	Water-level altitude 3/17/99 (feet above sea level)
P-250	26.4	27.13	10.4	15.0	14.57
<sup>1</sup> USGS-1	26.4	27.32	18.4	8.0	14.64
USGS-2	26.3	27.18	7.3	7.0	14.59
USGS-3	26.4	27.24	-5.6	5.0	15.72
P-251	28.4	29.27	--	--	14.63
P-254	29.5	30.84	--	--	16.34
P-601	26.6	26.96	--	--	13.30
P-602	25.8	26.41	--	--	13.39
P-603	26.4	27.23	--	--	13.35
<sup>1</sup> USGS-4	26.8	27.68	16.8	6.0	13.60
USGS-5	26.7	27.50	4.7	5.0	13.54
USGS-6	26.7	27.46	-5.3	8.0	13.77
P-604	26.6	27.33	11.6	25.0	13.21
P-605	26.8	28.11	--	--	15.81
P-606	27.1	27.58	--	--	15.81
P-607	27.1	27.66	--	--	15.49
P-608	27.2	27.86	--	--	15.12
P-609	26.2	26.80	11.2	25.0	12.33

<sup>1</sup> Does not yield sufficient water; no water sample collected.

**Table 3.** Results of analyses for volatile organic compounds in ground-water samples collected during February 8–10, 1999

[mg/L, milligrams per liter; µg/L, micrograms per liter; &lt;, less than; --, no data]

Well no. or identifier (fig. 3)	2-Chloroethanol (mg/L)	Ethane (µg/L)	Ethene (µg/L)	Bromoethane (µg/L)	Vinyl chloride (µg/L)	Chloroethane (µg/L)	Methylene chloride (µg/L)	Acetone (µg/L)	Carbon disulfide (µg/L)	1,1-Dichloroethene (µg/L)	<i>trans</i> -1,2-Dichloroethene (µg/L)	1,1-Dichloroethane (µg/L)	1,2-Dichloroethane (µg/L)	Chloroform (µg/L)
P-250	<1.0	<16	250	<200	7,284	<200	406	<200	<200	5,543	3,490	3,593	73,200	2,110
<sup>1</sup> USGS-2	<1.0	<16	260	<200	3,662	<200	<200	<200	<200	3,329	2,166	2,257	36,500	1,436
<sup>1</sup> USGS-3	<1.0	<16	140	<200	9,229	<200	216	<200	<200	13,250	7,239	7,456	209,200	5,094
P-251	<1.0	<16	96	<200	10,330	<200	<200	<200	<200	7,120	1,747	1,480	46,670	707
P-254	<1.0	<16	290	<200	3,110	<200	<200	<200	<200	1,638	576	532	7,019	352
P-601	<1.0	<16	110	<100	3,909	<100	<100	<100	<100	6,454	2,977	3,201	72,420	1,860
P-602	<1.0	<16	35	<100	2,510	<100	<100	<100	<100	4,304	2,012	2,170	40,810	1,223
P-603	<1.0	<16	<15	<100	325	<100	<100	<100	<100	1,047	315	309	2,087	183
<sup>1</sup> USGS-5	<1.0	<16	<15	<10	1,834	<10	<10	<10	<10	2,223	1,214	1,166	259	698
<sup>1</sup> USGS-6	<1.0	<16	<15	<10	<10	<10	<10	<10	<10	37	<10	<10	51	<10
P-604	<1.0	<16	<15	<100	114	<100	<100	<100	<100	516	109	<100	127	<100
P-605	<1.0	<16	320	<200	12,550	<200	334	1,036	<200	15,510	9,932	10,180	275,800	7,787
P-606	<1.0	<16	41	<20	2,591	<20	77	<20	<20	2,426	2,033	2,168	43,760	1,184
P-607	<1.0	<16	<15	<100	<100	<100	<100	<100	<100	<100	<100	<100	288	<100
P-608	<1.0	<16	<15	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10
P-609	<1.0	<16	<15	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10
Field blank	<1.0	<16	<15	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10
Trip blank	--	--	--	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10
Method blank	<1.0	<16	<15	--	--	--	--	--	--	--	--	--	--	--

Table 3

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Footnote at end of table.

**Table 3.** Results of analyses for volatile organic compounds in ground-water samples collected during February 8–10, 1999—Continued

Well no. or identifier (fig. 3)	2-Butanone (µg/L)	1,1,1-Trichloroethane (µg/L)	Carbon tetrachloride (µg/L)	Bromo-dichloromethane (µg/L)	1,2-Dichloropropane (µg/L)	<i>cis</i> -1,3-Dichloropropene (µg/L)	Tri-chloroethene (µg/L)	Benzene (µg/L)	<i>trans</i> -1,3-Dichloropropene (µg/L)	1,1,2-Trichloroethane (µg/L)	Di-bromo-chloro-methane (µg/L)	Bromo-form (µg/L)	4-Methyl-2-pentanone (µg/L)
P-250	<200	<200	<200	<200	<200	<200	655	290	<200	26,120	<200	<200	<200
<sup>1</sup> USGS-2	<200	<200	<200	<200	<200	<200	392	<200	<200	12,560	<200	<200	<200
<sup>1</sup> USGS-3	<200	<200	<200	<200	<200	<200	1,499	<200	<200	61,740	<200	<200	<200
P-251	<200	<200	<200	<200	<200	297	804	<200	327	32,780	<200	<200	<200
P-254	<200	<200	<200	<200	<200	<200	<200	<200	<200	3,530	<200	<200	<200
P-601	<100	<100	<100	<100	<100	<100	657	117	<100	25,760	<100	<100	<100
P-602	<100	<100	<100	<100	<100	<100	419	<100	<100	17,670	<100	<100	<100
P-603	<100	<100	<100	<100	<100	<100	<100	<100	<100	4,681	<100	<100	<100
<sup>1</sup> USGS-5	<10	<10	<10	<10	<10	<10	214	35	<10	5,308	<10	<10	<10
<sup>1</sup> USGS-6	<10	<10	<10	<10	<10	<10	<10	<10	<10	124	<10	<10	<10
P-604	<100	<100	<100	<100	<100	<100	<100	<100	<100	1,757	<100	<100	<100
P-605	<200	<200	<200	<200	<200	<200	2,459	<200	<200	73,950	<200	<200	<200
P-606	<20	<20	<20	<20	<20	<20	345	58	<20	8,843	<20	<20	<20
P-607	<100	<100	<100	<100	<100	<100	<100	<100	<100	179	<100	<100	<100
P-608	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10
P-609	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10
Field blank	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10
Trip blank	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10
Method blank	--	--	--	--	--	--	--	--	--	--	--	--	--

Footnote at end of table.

**Table 3.** Results of analyses for volatile organic compounds in ground-water samples collected during February 8–10, 1999—Continued

Well no. or identifier (fig. 3)	1,1,2,2-Tetrachloroethane (µg/L)	Tetrachloroethene (µg/L)	2-Hexanone (µg/L)	Toluene (µg/L)	Chlorobenzene (µg/L)	Ethylbenzene (µg/L)	Styrene (µg/L)	o-Xylene (µg/L)	m- and p-Xylene (µg/L)	cis-1,2-Dichloroethene (µg/L)	Chloromethane (µg/L)	Total xylenes (µg/L)	1,2-Dichloroethene, total (µg/L)
P-250	<200	<200	<200	223	266	<200	<200	217	<200	1,638	<200	217	4,204
<sup>1</sup> USGS-2	<200	<200	<200	<200	<200	<200	<200	<200	<200	877	<200	<200	3,042
<sup>1</sup> USGS-3	<200	351	<200	<200	491	<200	<200	<200	<200	2,714	<200	<200	9,773
P-251	289	<200	<200	441	542	261	<200	<200	<200	1,083	<200	<200	2,830
P-254	<200	<200	<200	<200	<200	<200	<200	<200	<200	328	<200	<200	904
P-601	<100	148	<100	<100	220	<100	<100	<100	<100	1,160	<100	<100	4,136
P-602	<100	<100	<100	<100	138	<100	<100	<100	<100	791	<100	<100	2,802
P-603	<100	<100	<100	<100	<100	<100	<100	<100	<100	175	<100	<100	490
<sup>1</sup> USGS-5	16	40	<10	<10	65	<10	<10	<10	<10	459	<10	<10	1,673
<sup>1</sup> USGS-6	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10
P-604	<100	<100	<100	<100	<100	<100	<100	<100	<100	<100	<100	<100	109
P-605	<200	396	<200	<200	592	<200	<200	<200	<200	3,941	<200	<200	13,300
P-606	<20	47	<20	<20	70	<20	<20	<20	<20	724	<20	<20	2,033
P-607	<100	<100	<100	<100	<100	<100	<100	<100	<100	<100	<100	<100	<100
P-608	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10
P-609	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10
Field blank	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10
Trip blank	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10	<10
Method blank	--	--	--	--	--	--	--	--	--	--	--	--	--

<sup>1</sup> See figure 1 for location.

**Table 4.** Results of analyses for field properties and constituents, inorganic ions, and dissolved organic carbon in ground-water samples collected during February 8–10, 1999[ $\mu\text{S}/\text{cm}$ , microsiemens per centimeter at 25 degrees Celsius;  $^{\circ}\text{C}$ , degrees Celsius;  $\text{mg}/\text{L}$ , milligrams per liter;  $\text{nM}/\text{L}$ , nanomoles per liter;  $\mu\text{M}/\text{L}$ , micromoles per liter; --, no data; <, less than]

Well no. or identifier (fig. 3)	Specific conductance ( $\mu\text{S}/\text{cm}$ )	pH	Water temperature ( $^{\circ}\text{C}$ )	Dissolved oxygen ( $\text{mg}/\text{L}$ )	Alkalinity as bicarbonate ( $\text{mg}/\text{L}$ )	Total iron species ( $\text{mg}/\text{L}$ )	Total iron II ( $\text{mg}/\text{L}$ )	Total sulfide ( $\text{mg}/\text{L}$ )	Molecular hydrogen ( $\text{nM}/\text{L}$ )	Methane ( $\mu\text{M}/\text{L}$ )	Dissolved chloride ( $\text{mg}/\text{L}$ )	Dissolved sulfate ( $\text{mg}/\text{L}$ )	Dissolved nitrite plus nitrate, as N ( $\text{mg}/\text{L}$ )	Dissolved organic carbon ( $\text{mg}/\text{L}$ )
P-250	--	--	--	--	--	--	--	--	--	--	352	61.4	<0.1	5.37
<sup>1</sup> USGS-2	2,270	7.05	22.2	0.04	670	0.41	0.3	0.02	2.59	18.5	382	123	<.1	5.29
<sup>1</sup> USGS-3	1,874	7.10	22.2	.07	573	.03	--	<.01	4.97	2.8	305	67.7	<.1	7.52
P-251	5,550	6.88	22.5	.13	516	.15	.15	.02	10.17	6.2	1,610	36.9	<.1	36.5
P-254	1,320	7.29	24.2	.28	514	.15	.04	.06	1.91	52.6	154	54.8	<.1	5.31
P-601	2,000	7.08	24.2	.11	639	.04	--	.07	1,312	17.5	303	93.6	<.1	5.3
P-602	1,749	--	23	--	--	--	--	--	326	11.4	269	83.8	<.1	2.22
P-603	1,577	7.46	23.7	.13	529	.02	--	.08	10.21	6.8	226	81.1	<.1	<1.0
<sup>1</sup> USGS-5	1,923	7.21	21.9	.10	583	.19	.12	.02	1.62	16	269	89.3	<.1	1.2
<sup>1</sup> USGS-6	1,500	7.43	26	.23	514	.18	.12	<.01	15.89	<.1	190	92.4	<.1	<1.0
P-604	1,689	7.40	21.6	.12	561	.08	--	.02	1.13	3.2	249	79.5	<.1	<1.0
P-605	1,995	7.15	23.1	.37	698	.07	--	--	4,765	1.1	339	64.6	<.1	14.2
P-606	1,531	7.64	22.1	--	547	.06	--	--	1,121	.8	199	80	<.1	<1.0
P-607	1,494	7.53	22.0	.10	646	.04	--	.01	13.33	.2	189	83.1	<.1	<1.0
P-608	1,370	7.63	21.5	.10	544	.13	.07	.03	1.01	<.1	158	53.5	<.1	<1.0
P-609	1,926	7.34	22.4	.10	567	.07	--	--	.06	<.1	328	88.4	<.1	<1.0
Field blank	--	--	--	--	--	--	--	--	--	--	<3.0	<2.0	<.1	<1.0
Method blank	--	--	--	--	--	--	--	--	--	--	<3.0	<2.0	<.1	<1.0

<sup>1</sup> See figure 1 for location.

**Table 5.** Simulated first-order decay constants for selected chlorinated ethenes and ethanes in ground water

Compound	First-order decay constant for upgradient segment of flowpath (per year)	First-order decay constant for downgradient segment of flowpath (per year)
Tetrachloroethene	<sup>1</sup> 0.46	0
Trichloroethene	.69	0
<i>cis</i> -1,2-Dichloroethene	.69	.10
<i>trans</i> -1,2-Dichloroethene	.50	.10
1,1-Dichloroethene	.50	0
Vinyl chloride	<sup>2</sup> .49	<sup>3</sup> .35
1,1,2-Trichloroethane	.50	0
1,2-Dichloroethane	.77	0
1,1-Dichloroethane	.45	.10

<sup>1</sup> Assumes tetrachloroethene concentrations identical in wells P-250 and P-601.

<sup>2</sup> Based on contribution of vinyl chloride from decay of *cis*-1,2-dichloroethene, *trans*-1,2-dichloroethene, and 1,1-dichloroethene.

<sup>3</sup> Based on contribution of vinyl chloride from decay of *cis*-1,2-dichloroethene and *trans*-1,2-dichloroethene.