USGS
science for a changing world

Rec'd
10/19/98

NATURAL ATTENUATION ASSESSMENT OF CONTAMINATED GROUND WATER AT A GAS-TURBINE MANUFACTURING PLANT, GREENVILLE, SOUTH CAROLINA

U.S. GEOLOGICAL SURVEY
Water-Resources Investigations Report 98-4165

Prepared in cooperation with the
SOUTH CAROLINA DEPARTMENT OF NATURAL RESOURCES-
WATER RESOURCES DIVISION

NATURAL ATTENUATION ASSESSMENT OF CONTAMINATED GROUND WATER AT A GAS-TURBINE MANUFACTURING PLANT, GREENVILLE, SOUTH CAROLINA

By Don A. Vroblesky, Matthew D. Petkewich, Paul M. Bradley,
and John F. Robertson

U.S. GEOLOGICAL SURVEY
Water-Resources Investigations Report 98-4165

Prepared in cooperation with the
SOUTH CAROLINA DEPARTMENT OF NATURAL RESOURCES-
WATER RESOURCES DIVISION

U.S. DEPARTMENT OF THE INTERIOR
MARCH 3, 1849

Columbia, South Carolina
1998

U.S. DEPARTMENT OF THE INTERIOR
BRUCE BABBITT, Secretary

U.S. GEOLOGICAL SURVEY
Thomas J. Casadevall, Acting Director

For additional information write to:

District Chief
U.S. Geological Survey
Stephenson Center-Suite 129
720 Gracern Road
Columbia, South Carolina 29210-7651

Copies of this report can be purchased from:

U.S. Geological Survey
Branch of Information Services
Box 25286
Denver, Colorado 80225-0286

CONTENTS

FIGURES

Figures 8 - 14. Maps showing:

TABLES

CONVERSION FACTORS, VERTICAL DATUM, AND ABBREVIATIONS

Multiply	By	To obtain
	Length	
inch (in.)	25.4	millimeter
foot (ft)	0.3048	meter
mile (mi)	1.609	kilometer
	Flow	
foot per day (ft/d)	0.3048	meter per day
foot squared per day (ft^2/d)	0.09290	meter squared per day
foot cubed per day (ft^3/d)	0.02832	meter cubed per day
gallons per minute (gal/min)	0.06308	liter per second
inches per year (in./yr)	25.4	millimeters per year
	Volume	
gallon (gal)	3.785	liter

Temperature: In this report, temperature is given in degrees Celsius (^oC), which can be converted to degrees Fahrenheit (^oF) by the following equation:

$$^oF = (9/5 \times {}^oC) + 32$$

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 the United States and Canada, formerly called Sea Level Datum of 1929.

Chemical concentration: In this report, chemical concentration in water is expressed in metric units as milligrams per liter (mg/L) or micrograms per liter (µg/L).

Biodegradation rate constant: In this report, first-order kinetics are used to describe the biodegradation rate of benzene. First-order rate constants (K) are expressed in units of percent of compound degraded per day (% d^{-1}). Because percent is dimensionless, this reduces to units of inverse days.

Other units used in report:

cubic feet per milligram	ft^3/mg
degrees Celsius	°C
disintegrations per minute	DPM
effective molecular diffusion coefficient	D_m
gram	g
liter	L
microgram	µg
micromole per liter	µM
micrometer	µm
milligram	mg
milliliter	mL
milliliter per minute	mL/min
moles per liter	M
nanomole per liter	nM
pounds per square inch	psi
ratio of horizontal transverse to longitudinal dispersivity	α_t/α_l

Abbreviations used in this report:

BTEX - benzene, toluene, ethylbenzene, and total xylenes
BTX - benzene, toluene, and total xylenes
CO_2 - carbon dioxide
$^{14}CO_2$ - carbon-14 labeled carbon dioxide
$[^{14}C]$ - carbon-14 labeled
CVOC - chlorinated volatile organic compound
DCA - 1,1-dichloroethane
DCE - *cis* 1,2-dichloroethene and *trans* 1,2-dichloroethene
DIC - dissolved inorganic carbon
DO - dissolved oxygen
DRO - diesel-range organic
Fe(II) - ferrous iron
Fe(III) - ferric iron
GC - gas chromatograph
H_2 - dissolved hydrogen
KOH - potassium hydroxide
MMOC - modified method of characteristics (solute-transport model solver)
MODFLOW - U.S. Geological Survey modular 3-D flow model
MT3D - modular 3-D transport model
NO_3 - nitrate
NPDES - National Pollutant Discharge Elimination System
PCE - tetrachloroethene
PCG2 - preconditioned conjugate gradient flow-model solver
PVC - polyvinyl chloride
RMSE - root mean squared error
TCA - 1,1,1-trichloroethane
TCE - trichloroethene
TEAP - terminal electron-accepting process
USEPA - U.S. Environmental Protection Agency
USGS - U.S. Geological Survey
VC - vinyl chloride
VLF-EM - very-low-frequency electromagnetic
VOC - volatile organic compound

NATURAL ATTENUATION ASSESSMENT OF CONTAMINATED GROUND WATER AT A GAS-TURBINE MANUFACTURING PLANT, GREENVILLE, SOUTH CAROLINA

By Don A. Vroblesky, Matthew D. Petkewich, Paul M. Bradley, *and* John F. Robertson

ABSTRACT

The potential for natural attenuation of petroleum-hydrocarbon and chlorinated volatile organic compound contamination in ground water was investigated at a gas-turbine manufacturing facility in Greenville, S.C. Examination of benzene migration involved integrating hydrogeologic, geochemical, and microbial parameters in a ground-water flow and solute-transport model. The natural attenuation potential of other compounds was examined using geochemical observations.

Modeling results suggest that natural attenuation processes probably will be adequate to prevent migration of benzene from the Test Stands contamination to Little Rocky Creek at concentrations greater than 5 µg/L. The potential for naphthalene and free-phase petroleum transport to the creek could not be determined because of limited historical data.

A substantial amount of chlorinated volatile organic compound dechlorination has taken place in the aquifer system. Most of the dechlorination appears to have taken place in zones where the chlorinated volatile organic compound contamination commingled with petroleum hydrocarbon contamination. Dechlorination in these zones of maximum degradation potential, however, is insufficient to prevent tetrachloroethene migration into downgradient areas where tetrachloroethene degradation is reduced. Thus, natural attenuation is not a suitable remedial alternative for contamination from the Diesel Pump Station or the Service Building.

By analogy, these data also provide a field experiment to test the potential of an engineered approach to optimize biodegradation rates in source areas by allowing the chlorinated aliphatic compounds to mix with oxidizable organic substrate. The data suggest that such an approach would be inadequate to completely prevent transport of tetrachloroethene in this aquifer system.

INTRODUCTION

The potential for naturally occurring attenuation processes to prevent transport of ground-water contaminants to property boundaries or points of contact with human or wildlife populations can be assessed by comparing rates of contaminant transport to rates of contaminant biodegradation and adsorption. If the rates of advective contaminant transport are fast relative to natural attenuation rates, the contaminants potentially can migrate to property boundaries or points of contact. If the advective contaminant transport rates are slow relative to attenuation rates, there is greater potential for reduction of contaminant concentrations prior to migration to property boundaries or points of contact.

Factors affecting contaminant transport and natural attenuation in saprolite and fractured-rock aquifers are not well understood. To examine these processes and their potential to reduce organic contaminant concentrations in saprolitic and fractured-bedrock aquifers, the U.S. Geological Survey (USGS), in cooperation with the South Carolina Department of Natural Resources, Water Resources Division, investigated ground-water contamination by petroleum hydrocarbons and chlorinated volatile organic compounds (CVOC) at a gas-turbine manufacturing facility in Greenville, S.C.

Purpose and Scope

The purpose of this report is to present the findings of an investigation to evaluate the potential for saprolite and fractured-rock aquifer processes to naturally attenuate ground-water contamination. Contaminants of interest include components of fuel-oil contamination in a saprolite/transition-zone aquifer and CVOC contamination in both a saprolite/transition-zone aquifer and a fractured-bedrock aquifer. The natural attenuation potential of selected fuel-oil components and CVOC's was qualitatively examined using hydrologic, water-chemistry, and microbial data. The potential for natural attenuation of benzene was analyzed more quantitatively by integrating hydrologic, geochemical, and microbial data into a digital solute-transport model to simulate contaminant movement to the year 2097.

Data from previous investigations as well as water-quality data collected during this investigation were used to evaluate the natural attenuation potential. Ground-water samples were collected from 20 wells (October 1995) and analyzed for major ions, dissolved oxygen (DO), selected minor constituents, and relevant organic contaminants.

Description of Study Area

The study area is a gas-turbine manufacturing plant in the Piedmont physiographic province near Greenville, S.C. (fig. 1). The ground-water system consists of three water-bearing zones: the saprolite, the underlying fractured bedrock, and a transition zone between the saprolite and the bedrock. For purposes of this investigation, the zones are grouped into two aquifers: the saprolite/transition-zone aquifer and the fractured-bedrock aquifer.

The site is characterized by ground-water contamination that originated from a variety of source areas. This report considers three areas of contamination: the Test Stands contamination, the Service Building contamination, and the Diesel Pump Station contamination. The types of ground-water contaminants include chlorinated solvents, such as tetrachloroethene (PCE), trichloroethene (TCE), and *cis* and *trans* 1,2-dichloroethene (DCE); free-phase diesel fuel (number 2 fuel oil); and dissolved petroleum hydrocarbons related to the diesel fuel, such as benzene, toluene, ethylbenzene, and total xylenes (collectively termed BTEX), diesel-range organics (DRO's), and naphthalene. The areal distribution of chlorinated solvents is distinct from the petroleum hydrocarbons in some areas and commingles with the petroleum hydrocarbons in other areas.

Observation wells in the study area include wells screened in the saprolite (designated with no alphanumeric suffix, as in WQ-05, or with the prefix "TP," as in TP-7) and others open to the shallow bedrock (designated with the suffix "B," as in WQ-27B). Wells designed to operate as contaminant-extraction wells are designated by the prefix "P," as in P-03. Wells having "A" as the suffix indicate that the screened interval is near the base of the saprolite or in the transition zone between saprolite and bedrock. In general, the well screens are 10-ft long. A series of wells with a "BW" prefix have 30-ft-long screens. The casing material is polyvinyl chloride (PVC). Construction data for observation wells in and adjacent to contaminated areas and selected background wells are shown in table 1.

Acknowledgments

Special thanks are extended to the site managers of the gas-turbine manufacturing facility for their cooperation. In addition, Kubal-Furr and Associates provided a water-chemistry data base that greatly assisted in this investigation.

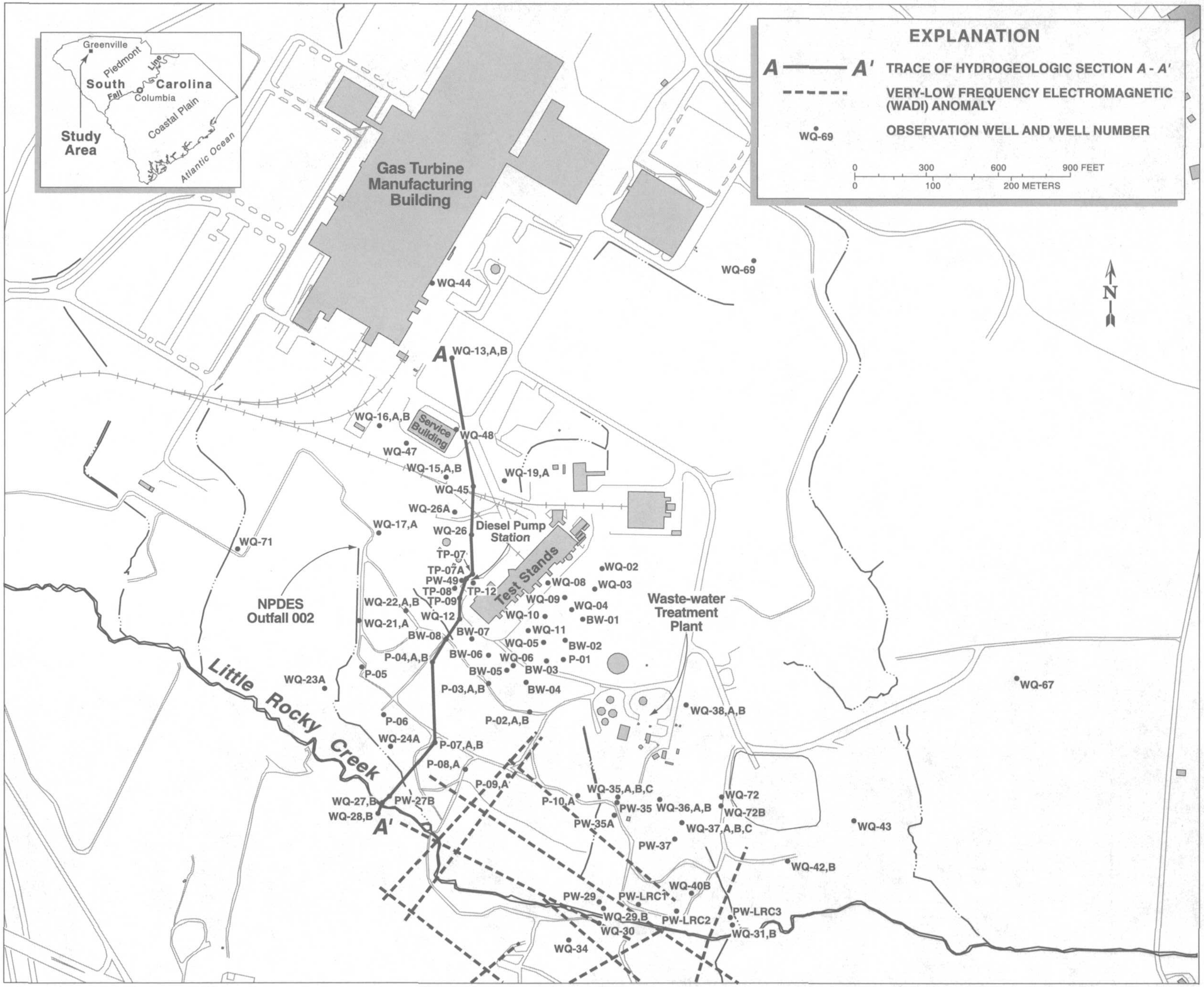


Figure 1. Location of study area, trace of hydrogeologic section A - A', observation-well locations, and very-low-frequency electromagnetic (WADI) anomalies at a gas-turbine manufacturing facility, Greenville, S.C.

Table 1. Construction data for selected observation wells at a gas-turbine manufacturing facility, Greenville, S.C.

[I.D., inside diameter; in., inches; ft, feet; lsd, land surface datum; sl, sea level]

Well number (fig. 1)	Casing I.D. (in.)	Well depth (ft below lsd)	Screen interval (ft below lsd)	Top of casing elevation (ft above sl)
BW-01	4	56.0	26.0-56.0	958.54
BW-02	4	59.5	29.5-59.5	959.01
BW-03	4	55.5	25.5-55.5	958.99
BW-04	4	65.5	35.5-65.5	958.67
BW-05	4	61	31-61	958.54
BW-06	4	56	26-56	958.24
BW-07	4	50	20-50	955.77
BW-08	4	39.3	9.3-39.3	944.51
P-01	2	40.89	30.89-40.89	959.39
P-02	2	34.54	24.54-34.54	938.43
P-03	2	26.55	16.55-26.55	939.22
P-04	2	24.48	14.48-24.48	946.04
P-04A	2	60.55	50.55-60.55	945.67
P-04B	2	80.59	70.59-80.59	945.53
P-07	2	15.59	5.59-15.59	933.75
P-07A	2	26.1	16.10-26.10	933.94
P-07B	2	54	44.00-54.00	933.78
PW-27B	6	62.1	48.50-62.10	924.43
TP-07	2	24.49	14.49-24.49	959.92
TP-07A	2	79.21	69.2-79.2	960.14
TP-08	2	34.26	24.26-34.26	959.83
TP-09	2	33.98	23.98-33.98	960.59
TP-12	2	27.75	17.75-27.75	960.83
WQ-01A	2	68.96	58.96-68.96	951.28
WQ-02	2	24.3	14.3-24.3	949.48
WQ-03	2	33.35	23.35-33.35	959.77
WQ-04	2	36.65	26.65-36.65	960.84
WQ-05	2	40.6	30.6-40.6	960.36
WQ-06	2	47.52	37.52-47.52	959.94

Table 1. Construction data for selected observation wells at a gas-turbine manufacturing facility, Greenville, S.C. --Continued
[I.D., inside diameter; in., inches; ft, feet; lsd, land surface datum; sl, sea level]

Well number (fig. 1)	Casing I.D. (in.)	Well depth (ft below lsd)	Screen interval (ft below lsd)	Top of casing elevation (ft above sl)
WQ-08	2	32.38	22.38-32.38	962.09
WQ-09	2	42.76	32.76-42.76	961.53
WQ-10	2	42.48	32.48-42.48	961.39
WQ-11	2	48.25	38.25-48.25	961.09
WQ-12	2	30.43	20.43-30.43	960.21
WQ-13	2	28.3	16.93-26.43	963.97
WQ-13A	2	61.65	51.65-61.65	964.09
WQ-13B	4	92.4	73.00-92.40	963.78
WQ-15	2	47.28	37.28-47.28	959.81
WQ-15A	2	65.94	55.94-65.94	960.47
WQ-15B	2	92.69	82.69-92.69	960.09
WQ-17	2	27.94	17.94-27.94	959.99
WQ-21	2	13.93	3.93-13.93	950.58
WQ-22	2	13.97	3.97-13.97	952.69
WQ-26	2	34.2	23.85-34.1	960.02
WQ-26A	2	78.84	68.84-78.84	960.76
WQ-27	2	16.25	4.52-13.85	923.77
WQ-27B	4	58.87	39.00-58.87	924.02
WQ-28	2	19	8.31-17.66	924.63
WQ-28B	4	61	39.60-61.00	925.07
WQ-45	2	23.57	8.8-23.57	961.50

FIELD AND LABORATORY METHODS FOR ASSESSMENT OF CONTAMINANT ATTENUATION

Established methods were used in this investigation to assess attenuation of chlorinated hydrocarbons in a fractured-rock aquifer and petroleum hydrocarbons in a saprolite/transition-zone aquifer. In general, the approach was to compare advective contaminant transport processes to the factors affecting contaminant attenuation. Specific methods used during this investigation are discussed in the following sections.

Ground-Water Sample Collection and Analysis

All monitoring wells were purged of casing water prior to collecting water samples. At least three casing volumes of water were removed from each well by using either a Teflon bailer or a positive-displacement pump. Ferrous iron [Fe(II)], dissolved oxygen (DO), pH, and temperature in ground water were measured in the field.

After well purging, the DO concentration in water from each well was determined by Winkler titration (Hach Company, 1983). Ferrous iron concentration was measured using the Hach colorimeter/ FerroZine method (Stookey, 1970). The temperature and pH were measured on a Beckman digital pH meter.

Water samples for analysis of volatile organic compounds (VOC's) and DRO's were collected at selected wells by Rust Environment and Infrastructure/ Kubal Furr and Associates following established U.S. Environmental Protection Agency (USEPA) protocols (Kubal-Furr and Associates, 1996). All sample bottles were capped with Teflon-lined bottle caps. The water samples for VOC's and DRO's were analyzed using USEPA methods 8240 and 3550/8015, respectively (U.S. Environmental Protection Agency, 1986).

To determine methane and dissolved inorganic carbon (DIC) concentrations, 5 milliliters (mL) of sample water were collected using a clean syringe and injected into septated vials through a 0.45-micrometer (µm) porous-membrane filter. The vials were rinsed with filtered sample water prior to sampling. The samples were stored on ice until analyzed. Methane was quantified by thermal-conductivity-detection gas chromatography. Dissolved methane concentrations were calculated using Henry's Law coefficients (Stumm and Morgan, 1981). The DIC samples were acidified in the laboratory with a 42.5 percent phosphoric acid solution, and DIC concentrations were quantified by thermal-conductivity-detection gas chromatography. Sample DIC concentrations were corrected for carbon dioxide (CO_2) present in sample-free blanks collected in the field.

Water samples for analysis of inorganic ions were collected in sample-rinsed polyethylene bottles after passing through a 0.45-µm porous-membrane filter. The inorganic ion samples were packed in ice immediately following collection. The samples were analyzed for calcium (method 215.1), magnesium (method 242.1), sulfate (method 375.4), and nitrate (method 352.1) (U.S. Environmental Protection Agency, 1983). Anions were analyzed using ion-exchange chromatography with chemical suppression [method 300.0 (U.S. Environmental Protection Agency, 1986)].

The use of dissolved hydrogen gas (H_2), a transient intermediate product of anaerobic microbial metabolism, as an indicator of predominant terminal electron-accepting processes (TEAP's) was introduced by Lovley and Goodwin (1988). This approach has been used to document the zonation of TEAP's in various systems by Chapelle and Lovley (1990; 1992), Chapelle and McMahon (1991), and Vroblesky and Chapelle (1994). These studies concluded that characteristic dissolved hydrogen concentrations were 5 to 25 nanomoles per liter (nM) for methanogenesis; 1 to 4 nM for sulfate reduction; and 0.1 to 0.8 nM for ferric iron [Fe(III)] reduction. Because hydrogen is an extremely transitory intermediate, with a half-life of less than a minute (Conrad and others, 1987), it is a useful constituent for characterizing the local predominant TEAP's.

A ground-water hydrogen sample was collected from well WQ-15B using the bubble-strip method of Chapelle and McMahon (1991). A stream of water was pumped from the well through a gas-sampling bulb at an approximate rate of 600 mL per minute (mL/min). A bubble of nitrogen was injected into the bulb. Hydrogen and other soluble gases partitioned from the water into the bubble until equilibrium was achieved. Once equilibrium was achieved (less than 5 percent change in hydrogen concentration in 5 minutes, which typically occurred within 30 minutes of initiating the flow of water through the bulb), gas was extracted from the bulb using a gas-tight syringe. Dissolved hydrogen was measured using a gas chromatograph (GC) equipped with a reduction gas detector. The detection limit of this method varied between 0.1 and 0.5 nM. All dissolved hydrogen samples were collected as duplicates.

Compilation of Water-Level Data

Quarterly water-level data, collected during previous investigations, were used in this investigation to determine field-derived average-annual water levels for non-pumping and pumping conditions. Field-derived average-annual water levels for the non-pumping period were determined by compiling quarterly water-level data for April 1991 through April 1994. Spring, summer, fall, and winter water-level data from individual wells were separately averaged to determine average seasonal water levels, which then were averaged to yield field-derived average-annual water levels for each well. The field-derived average-annual water levels for the pumping period were determined in the same manner as the non-pumping period by compiling quarterly data for June 1994 through July 1997. This allowed approximately 2 months for the pumping stresses to achieve hydrodynamic stability. A pumping test in the saprolite near the Test Stands showed that the aquifer reached hydrodynamic stability within 23 hours (Kessler and Cross, 1986). Thus, the compiled water levels represent approximate steady-state conditions.

At wells where measured water levels were available only for a limited number of annual quarters, the average water levels were determined for the available time periods, or the period-of-interest. These values were adjusted to approximate field-derived average-annual water levels by a statistical comparison to wells where field-derived average-annual data were available. Field-derived average-annual water levels were graphed relative to average period-of-interest water levels at wells where data were available for all four annual quarters. The linear regression of this relation (coefficient of determination R^2 was 0.99) provided an equation that was used to adjust the average period-of-interest water levels to approximate field-derived average-annual water levels at wells with limited data. This methodology was used to approximate separate field-derived average-annual water levels for non-pumping and pumping periods at wells where measured water levels were missing for some of the annual quarters.

Quantification of Biodegradation Rates

The ability of microorganisms indigenous to the aquifer system underlying the site to mineralize specific compounds under aerobic and anaerobic conditions was evaluated using both radiolabeled and nonlabeled compounds. Sediment samples were collected from the saprolite below the water table using a flame-sterilized split-spoon sampler through hollow-stem augers. In addition, a sample of auger cuttings was obtained from near well WQ-10 (fig. 1) for determination of selected petroleum-hydrocarbon biodegradation rates. Samples were maintained at 4 °C until microcosm studies were initiated.

Petroleum-Hydrocarbon Biodegradation Rates

Biodegradation rates of benzene and decane under aerobic conditions were evaluated in aquifer-sediment microcosms using uniformly ring-labeled carbon-14 [^{14}C] benzene and uniformly labeled [^{14}C] decane. Microcosm experiments were initiated using saprolite sediments collected adjacent to wells and from a background area approximately 1,600 ft northeast of the Test Stands contamination area.

Microcosms consisted of 20 mL serum vials which were amended with 10 grams (g) of aquifer sediment and sealed with Teflon-lined butyl rubber stopper/base trap assemblies (Bradley and Chapelle, 1996). Microcosms were created with a head-space of air and amended with 0.5 mL of anoxic, sterile distilled water. Killed controls were prepared as described and autoclaved twice for 1 hour at 15 pounds per square inch (psi) and 121 °C. Three live and duplicate killed controls were prepared for each substrate treatment and each sediment (split-spoon samples from near well WQ-11 and from the background area, and auger cuttings from near well WQ-10).

The microcosms were incubated for a 2-day acclimation period. Then, approximately 200,000 disintegrations per minute (DPM) of [^{14}C] benzene or [^{14}C] decane were injected directly into the saturated sediment to yield a final dissolved concentration of about 1 micromole per liter (μM). The purity of the radiolabeled substrates was determined by radiometric detection gas chromatography to be greater than 98 percent.

To sample, microcosm base traps were rinsed with 0.5 mL of sterile distilled water and filled with 0.3 mL of 3 moles per liter (M) potassium hydroxide (KOH). After 12 hours, the KOH solution was removed and the amount of trapped carbon-14 labeled carbon dioxide ($^{14}CO_2$) was quantified using scintillation counting. Production of $^{14}CO_2$ was confirmed in selected vials by addition of barium chloride as described previously (Davis and Carpenter, 1990). The fact that no radioactivity was detected in the base traps of blanks consisting of sterile serum vials that contained radiolabeled benzene or decane but no sediment indicates that trapping of radiolabeled benzene or decane was not significant (less than 0.5 percent) in experimental microcosms. Microcosms were incubated in the dark at room temperature, and mineralization was monitored periodically over 20 days.

Rates of [^{14}C] benzene and decane mineralization were estimated by linearly regressing the percentage of ^{14}C recovered as $^{14}CO_2$ in base traps as a function of incubation time using a simple linear regression program (Sigmastat, Jandel Scientific, San Rafael, Calif.). Rates of [^{14}C] benzene and decane mineralization were considered insignificant in treatments where the final recovery of ^{14}C was less than 2 percent (the purity of the [^{14}C] substrates was 98.5 percent) or the slope of the linear regression was not statistically significant from zero (*P* value less than 0.05, Sigmastat, Jandel Scientific, San Rafael, Calif.).

Tetrachloroethene Biodegradation Rates

The ability of indigenous, aquifer microorganisms to degrade PCE under aerobic and anaerobic conditions also was evaluated. Sediment samples near well P-04 were collected using a flame-sterilized split-spoon sampler and stored in autoclaved jars. All samples were maintained at 4 °C until microcosm studies were initiated.

Microcosms consisted of 60 mL serum vials that were amended with 50 g of aquifer sediment and sealed with Teflon-lined butyl-rubber stoppers. Microcosms were created with a head-space of air (aerobic treatments) or helium (anaerobic treatments) and amended with 20 mL of anoxic, sterilized ground water collected from well P-04. Killed controls were prepared as described and autoclaved twice for 1 hour at 15 psi and 121 °C. Three live and duplicate killed controls were prepared for each head-space treatment (aerobic or anaerobic) and each aquifer material. The microcosms were incubated for a 2-day acclimation period and then amended with PCE to yield a dissolved concentration of approximately 700 micrograms per liter (μg/L). Microcosms were shaken to thoroughly mix the PCE with the water and sediment and allowed to equilibrate for 12 hours. Microcosms were incubated in the dark, and the head-space concentrations of PCE and its daughter products TCE, DCE, vinyl chloride (VC), and ethene were monitored periodically over 100 days.

Quantification of Adsorption

Adsorption coefficients for benzene and decane were determined using a modification of the method described in Schwarzenbach and Westall (1981). Approximately 5 g of aquifer material collected from a boring near well P-04 were placed in 10 mL serum vials. The vials were filled with solutions containing uniformly labeled [^{14}C] benzene or uniformly labeled [^{14}C] decane at concentrations of 0.2 to 100 μg/L benzene or 0.2 to 5 μg/L decane. Vials were sealed without airspace using Teflon-coated butyl rubber stoppers. Vials were prepared in duplicate for each compound at each test concentration. The final volume of solution in the adsorption vials was 7.4 ± 0.1 mL. Adsorption vials were vigorously shaken for 5 minutes and allowed to stand overnight.

Final dissolved concentrations of [^{14}C] benzene and [^{14}C] decane were determined by removing 1 mL of solution from each vial and quantifying radiolabelled solute by liquid scintillation counting. The difference between final dissolved concentrations observed in adsorption vials and the concentration in the source solutions was attributed to adsorption.

HYDROGEOLOGY

Ground water in the study area moves through zones of saprolite, underlying fractured metamorphic rock, and a transition zone in between these two units. The saprolite primarily consists of silt and fine- to medium-grained silty sand, and the bedrock primarily consists of biotite gneiss, although granite gneiss also has been reported (Koch, 1968; Sirrine Environmental Consultants, 1992). The saprolite thickness varies from 0 ft where bedrock crops out in Little Rocky Creek to approximately 100 ft in upgradient areas (Davis and Floyd, 1994). The approximate altitude of the bedrock ranges from about 840 to 910 ft, which was determined during previous investigations by initially contouring the depth of auger refusal (SEC Donohue, 1992a) and subsequently adjusting for probable differences in that depth caused by differing drilling methodologies (Stacy Sargent, Rust Environment and Infrastructure, Inc., written commun., 1996). The approximate saturated thickness of the saprolite/transition-zone aquifer determined in this investigation was 33 to 67 ft (fig. 2), derived by subtracting the altitude of the discretized bedrock surface (Stacy Sargent, Rust Environment and Infrastructure, Inc., written commun., 1996) from the discretized surface of the approximate mean annual water table (fig. 3). The discretized surface of the approximate mean annual water table used for this calculation was derived from model simulations for this investigation and will be discussed later in this report.

Hydrogeologic Framework

Ground-water recharges by infiltration of local precipitation. Normal rainfall near Greenville, is 51.27 in. per year (in./yr) (National Oceanic and Atmospheric Administration, 1996). Recharge rates were not measured for this study; however, measured recharge rates from 12 sites in regolith/fractured rock in Orange

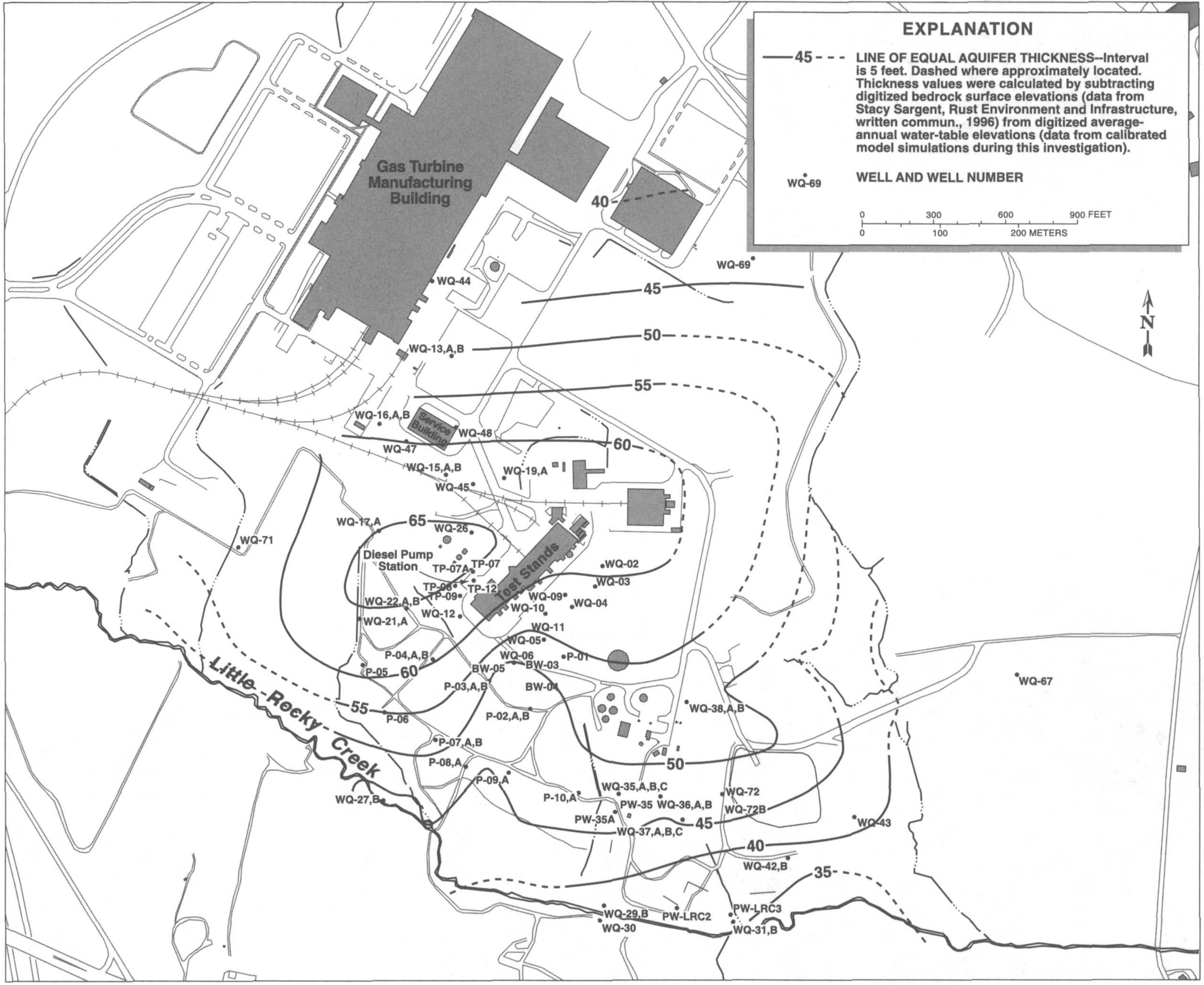


Figure 2. Approximate saturated thickness of the saprolite/transition-zone aquifer at a gas-turbine manufacturing facility, Greenville, S.C.

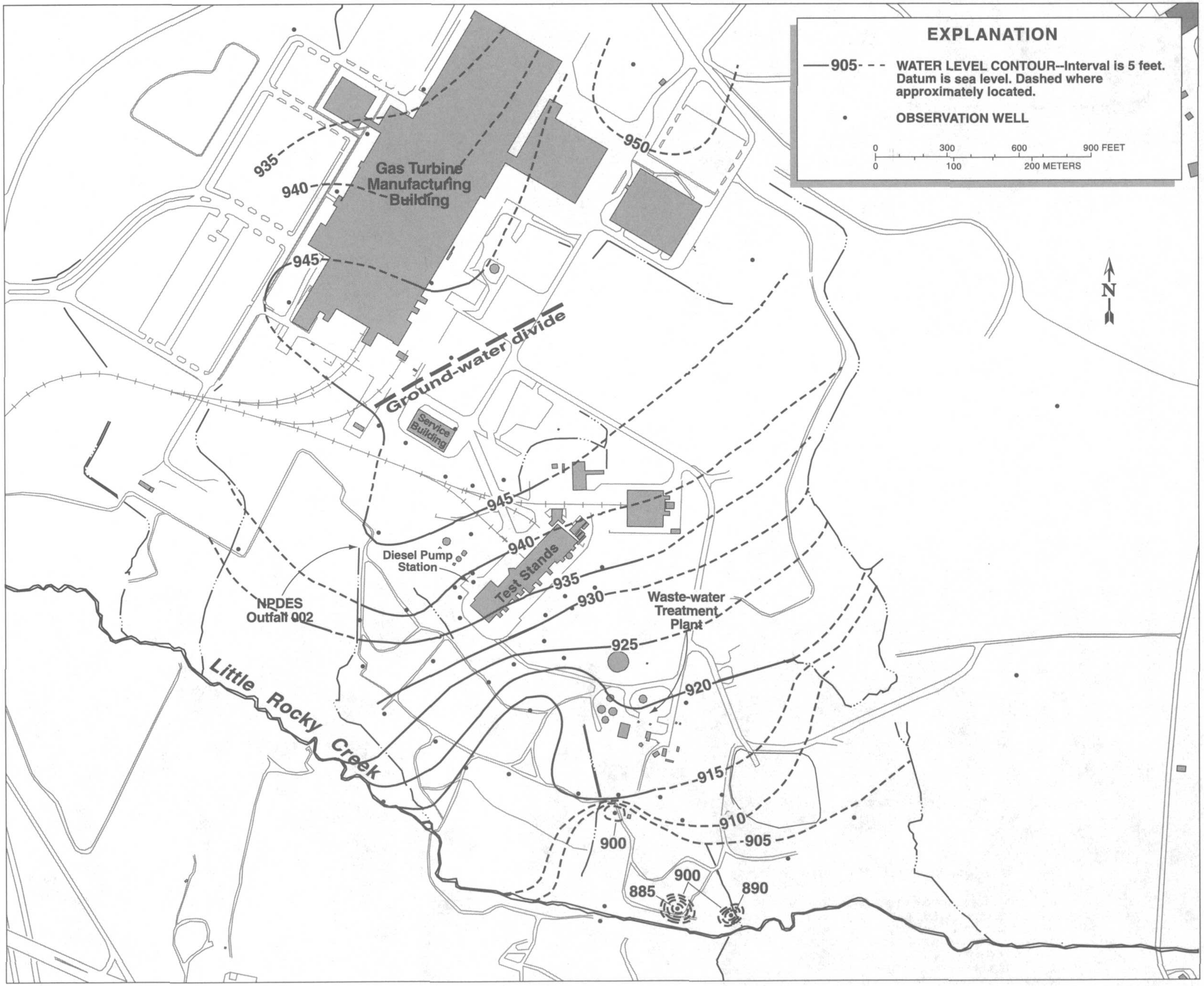


Figure 3. Average annual water levels under pumping conditions at a gas-turbine manufacturing facility, Greenville, S.C., June 1994 through July 1997.

County, North Carolina, ranged from 4.15 to 6.40 in./yr, with a mean of 4.90 in./yr (Daniel, 1996). These values probably are similar to those in the study area.

During June 1994 to July 1997, when ground-water pumping was in operation, the field-derived average-annual water levels at several wells (such as wells P-03, P-04, TP-07, TP-08, TP-12, WQ-06, WQ-10, WQ-43, and others) distant from pumping centers were higher than they were during the non-pumping conditions. These data imply that recharge to the aquifer was slightly higher during the pumping period than the non-pumping period. This hypothesis is supported by the rainfall data from the Greenville/Spartanburg Airport (National Oceanic and Atmospheric Administration, 1996), showing that the average monthly rainfall was 4.03 in. (standard deviation of 2.4 for 25 months) during April 1991 to April 1994, prior to operation of the pumping wells, and was 4.61 in. (standard deviation of 2.8 for 40 months) during the pumping period from June 1994 to July 1997.

Ground-water discharge is dominantly to Little Rocky Creek and its tributaries (fig. 3). During November 1991, the average stream-flow increase across the reach of Little Rocky Creek approximating the reach addressed by this investigation was 57 gal/min, or 11,000 ft^3/d (SEC Donohue Inc., 1992a). This value was reported as a mixture of baseflow and surface runoff. Ground water also discharges to a series of wells that began operation in April 1994 south of the wastewater-treatment plant to pump and remove ground-water contamination emanating from unlined lagoons (fig. 3).

Horizontal and vertical hydraulic conductivity values in the saprolite and horizontal hydraulic conductivity values in bedrock were determined during a previous investigation (SEC Donohue, 1992a). In the southeastern part of the modeled area, slug test results at wells WQ-42 and WQ-43 (fig. 1) showed that the hydraulic conductivity values were approximately 0.97 and 1.7 ft/d, respectively. A slug test at well WQ-38 showed the hydraulic conductivity to be about 7.7 ft/d. Hydraulic conductivities were measured at well WQ-29 (2.9 ft/d) and on the opposite side of Little Rocky Creek at wells WQ-30 (15 ft/d) and WQ-34 (3.2 ft/d). The geometric mean of horizontal hydraulic conductivity in the saprolite was about 2.6 ft/d, based on slug tests. Vertical hydraulic conductivity values determined by laboratory testing of undisturbed cores at wells WQ-42, WQ-45, and WQ-40B (fig 1) were 1.4, 2.0×10^{-2}, and 5.6×10^{-4} ft/d, respectively, with a geometric mean of 0.025 ft/d. The data indicated an anisotropic distribution of hydraulic conductivity, with the horizontal hydraulic conductivity being approximately 100 times higher than the vertical hydraulic conductivity (SEC Donohue, 1992a). The horizontal hydraulic conductivity of fracture zones in the bedrock (from 0.01 to 5.0 ft/d with a geometric mean of 0.7 ft/d), determined from aquifer tests, was lower than in the saprolite (SEC Donohue, 1992a).

Rising-head aquifer tests were done during a previous investigation to compare hydraulic conductivity in the saprolite and transition zone (SEC Donohue, 1992a). Calculated hydraulic conductivities in the saprolite and transition zone were 0.6 and 0.4 ft/d, respectively, at well-cluster WQ-35; 10 and 0.4 ft/d, respectively, at well-cluster WQ-36; 0.6 and 1.2 ft/d, respectively, at well-cluster WQ-37; and 7.7 and 5.0 ft/d, respectively, at well-cluster WQ-38 (fig. 1). Thus, the data from most well clusters show a similarity in reported hydraulic conductivities in the two units.

Although water-levels in upgradient wells generally are slightly higher in the shallow saprolite than in the deeper saprolite or transition zone, the difference is small, typically 0.7 ft or less. These data, and the observation that horizontal hydraulic conductivity is approximately 100 times larger than the vertical hydraulic conductivity (SEC Donohue, 1992a), imply that the direction of flow in the shallow saprolite in upgradient areas is approximately horizontal (Davis and Floyd, 1994).

Ground water in the vicinity of the Service Building and the Test Stands moves generally southwardly away from a ground-water divide oriented from northwest to southeast between the Service Building and the Manufacturing Building (Davis and Floyd, 1994; Kubal-Furr and Associates, 1996) (fig. 3). Shallow ground water moves toward Little Rocky Creek and radially toward a zone of low ground-water levels (fig. 3) near well P-02 (fig. 1).

The location and inferred orientation of the low ground-water-level zone near well P-02 are in the approximate area of two very-low-frequency electromagnetic (VLF-EM) anomalies identified during a WADI survey (Sirrine Environmental Consultants, 1991a; Vroblesky, Rhodes, and others, 1996) (fig. 1). The VLF-EM anomalies were linear, trending southwestward from the vicinity of well P-02 to Little Rocky Creek (fig. 1).

The VLF-EM anomalies detected by the WADI survey arise from secondary electromagnetic fields generated when subsurface VLF signals from various military radio transmissions around the world encounter a low resistivity zone, such as a water-filled fracture in host rock. The distribution of these secondary fields, as detected by a WADI survey, are interpreted to represent water-filled fracture zones at the facility (Sirrine Environmental Consultants, 1991a; Vroblesky, Rhodes, and others, 1996). Measurements of water levels in deeper wells (wells P-02A and P-02B), immediately adjacent to well P-02, show a net upward hydraulic gradient from the fractured-rock aquifer to the saprolite (SEC Donohue, 1992a), indicating that the ground-water depression is not the result of downward gradient and flow. Thus, the probable explanation for the ground-water depression near well P-02 is that it represents a zone of anomalously high hydraulic conductivity and a preferential pathway for lateral water movement from the area near well P-02 toward Little Rocky Creek. Shallow test borings near well WQ-09, upgradient from well P-02, showed the presence of a buried alluvial channel roughly aligned in the direction of the ground-water depression (Kessler and Cross, 1986). Thus, the zone may indicate the presence of a paleochannel or fracture zone in the saprolite/transition-zone aquifer.

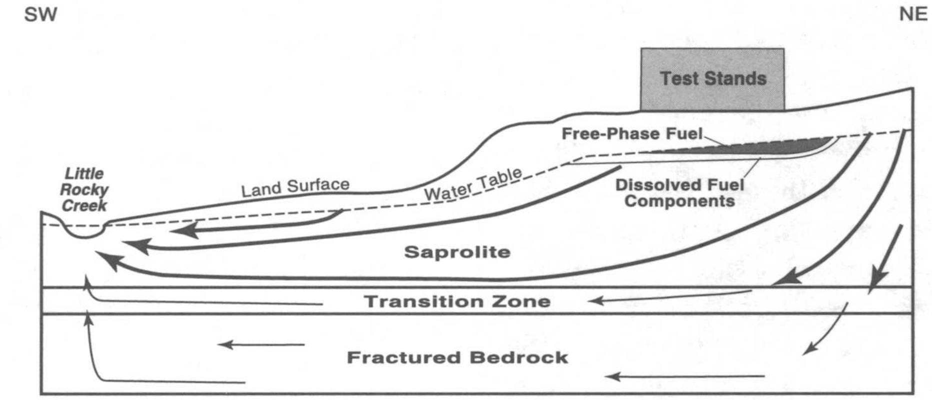
Little Rocky Creek and its tributaries are the areas of shallow ground-water discharge, as evidenced by upward hydraulic gradients in well clusters near the creek (Kubal-Furr and Associates, 1996). During November 1991, the average stream-flow increase from baseflow and surface runoff to Little Rocky Creek within the boundaries of the manufacturing plant was 57 gal/min, or 11,000 ft^3/d (SEC Donohue Inc., 1992a).

The creek flows from west to east along the approximate southern boundary of the facility. The creek-bottom sediment is predominantly silty sand, but locally contains outcropping rock. The creek ranges in width from about 2 ft in areas where flow is constricted by exposed rock to about 14 ft. The depth of water in the thalweg ranges from about 0.2 to 4 ft. The creek intercepts shallow ground-water flow, but deeper ground water of the fractured-rock aquifer locally appears to flow beneath the creek, as indicated by the transport of contaminants beneath the creek along certain reaches (Vroblesky and others, 1997).

Conceptual Model of Ground-Water Flow

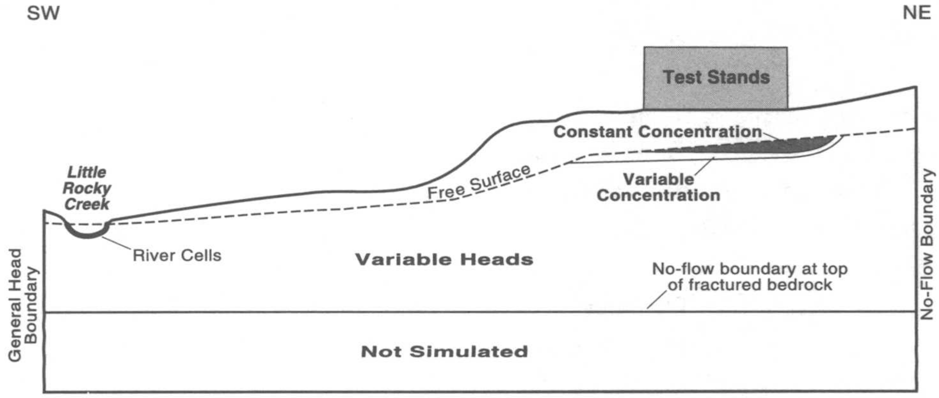
The ground-water system can be thought of as containing three water-bearing zones: the saprolite, the underlying fractured bedrock, and a transition zone between the saprolite and the bedrock (fig. 4). Rainfall infiltration recharges the saprolite/transition-zone aquifer, and most of the recharged water in the study area discharges to Little Rocky Creek and its tributaries. Part of the recharged water is discharged to the atmosphere by evapotranspiration, and a relatively small portion of the water in upgradient areas enters the underlying fractured bedrock. A series of ground-water contamination recovery wells approximately 1,000 ft southeast of the Test Stands have been in operation since May 1994 and provide an additional ground-water sink. In addition to describing ground-water flow, this conceptualization is useful for discussions of CVOC's in the aquifer because, as will be shown, density gradients have transported the CVOC's downward to each zone. This conceptualization also was used in a previous ground-water flow model of a large area, approximately 2.4 mi east to west and approximately 5.9 mi north to south, with the Test Stands in the approximate center of the modeled area (Stacy Sargent, Rust Environment and Infrastructure, written commun., 1996). The earlier simulations were used to evaluate the hydraulic effects of selected remediation alternatives in various areas of the facility (Rust Environment & Infrastructure, 1995b).

For the purpose of simulating benzene solute transport in this investigation, the ground-water flow system is further simplified to one layer, the saprolite/transition-zone aquifer. Simulating the ground-water flow as a one-layer system is justified for several reasons. As will be discussed, it is highly probable that benzene in the contamination plume immediately southeast of the Test Stands is limited to the saprolite/transition-zone aquifer. This consideration, as well as the larger average horizontal hydraulic conductivity of the saprolite (2.6 ft/d) relative to the average horizontal hydraulic conductivity of the fractured bedrock (0.7 ft/d), and measured vertical anisotropy (SEC Donohue, 1992a) justifies limiting the simulations to horizons shallower than the bedrock.



GENERALIZED HYDROGEOLOGIC SECTION (Not to scale)

NOTE: Arrows indicate general direction of ground-water flow. Bolder arrows indicate relatively greater flow.



GROUND-WATER-FLOW AND SOLUTE-TRANSPORT MODEL CONCEPTUALIZATION IN GENERALIZED HYDROGEOLOGIC SECTION

Figure 4. Conceptualization of ground-water flow and solute transport for model simulation in the saprolite/transition-zone aquifer at a gas-turbine manufacturing facility, Greenville, S.C.

Further justification can be found by examining water-level data. Hydraulic gradients indicate that water discharges from the saprolite into the underlying fractured bedrock in some upgradient areas; however, such flow appears to be small relative to lateral flow in the saprolite (SEC Donohue, 1992a). In downgradient areas, ground water appears to discharge from the bedrock upward into the saprolite; however, part of the ground water in the fractured bedrock flows under local discharge zones, such as Little Rocky Creek, and continues to move toward offsite, regional discharge zones (Vroblesky and others, 1997). The data imply that most of the ground water that enters the fractured bedrock from the saprolite in the simulated area returns to the saprolite or to Little Rocky Creek. Because the fractured bedrock does not appear to constitute a major sink or source for saprolite ground water in the modeled area, exclusion of the fractured bedrock from the digital model should not substantially affect the model solution.

Similarity in hydraulic conductivity values justifies combining the saprolite and the transition zone into a single unit for simulation purposes. Although the transition zone elsewhere has been reported to be somewhat more permeable than the overlying regolith (Stewart, 1962; Nutter and Otton, 1969), and the transition zone near the wastewater-treatment plant (fig. 1) is a transport pathway for chlorinated solvents (W.T. Hyde, General Electric Company, oral commun., 1998), rising-head aquifer tests in this study area show the transition zone hydraulic conductivity to be similar to or, in some cases, slightly lower than the overlying saprolite hydraulic conductivity (SEC Donohue, 1992a). These data suggest that the saprolite/transition-zone aquifer is the predominant pathway of benzene-contaminated ground-water flow immediately southeast of the Test Stands.

GROUND-WATER CONTAMINATION

Three areas of ground-water contamination are considered in this report. The areas, which overlap to some extent, are the Test Stands contamination, the Service Building contamination, and the Diesel Pump Station contamination.

Test Stands Contamination

The Test Stands contamination (fig. 5) originated in January 1984 from a leak of approximately 16,000 gal of diesel fuel from an underground pipe at the northwest corner of the Test Stands building. The contamination has subsequently spread south and east of the building. Dissolved components of the fuel include BTEX compounds and naphthalene. A system of extraction wells to recover fuel was installed downgradient from the contamination and operated from 1986 until March 1991. Approximately 4,800 to 5,200 gal of free-phase fuel were recovered by various methods (Kubal-Furr and Associates, 1996). In addition, 1,1,1-trichloroethane (TCA) may have been released into a storm drain at the Test Stands that discharged onto land surface (SEC Donohue, 1992a). Subsequent testing found TCA in sediment samples.

Distribution

Early investigations showed that the free-phase fuel was present at the northern part of the Test Stands in a zone extending at least 50 ft west of the building to at least 30 ft east of the building (Law Engineering Testing Company, 1984a; 1984b). Free-phase fuel is present at two wells (WQ-08 and WQ-11) in the Test Stands contamination, and dissolved DRO's are present in ground water at several nearby wells (fig. 5). Specific dissolved components of fuel present downgradient from the free-phase fuel include BTEX and naphthalene.

The history of Test Stands contaminant concentrations cannot be determined precisely for the fuel-line leak because petroleum hydrocarbon contamination also entered the aquifer in the vicinity of the Diesel Pump Station, and the two plumes probably commingle in places. Moreover, the contamination was mapped by several investigators using a variety of analytical methods. The Test Stands contamination was delineated using the summation of BTEX or benzene, toluene, and total xylenes (BTX) concentrations prior to the second quarter of 1992 (SEC Donohue, 1992b). After 1992, site investigators ceased sampling for BTX and BTEX and began characterizing the contamination using DRO concentrations. In 1995, site investigators began characterizing the contamination using an analysis of DRO's that apparently summed a different suite of hydrocarbons than in previous DRO analyses (W.T. Hyde, General Electric Company, oral commun., 1995).

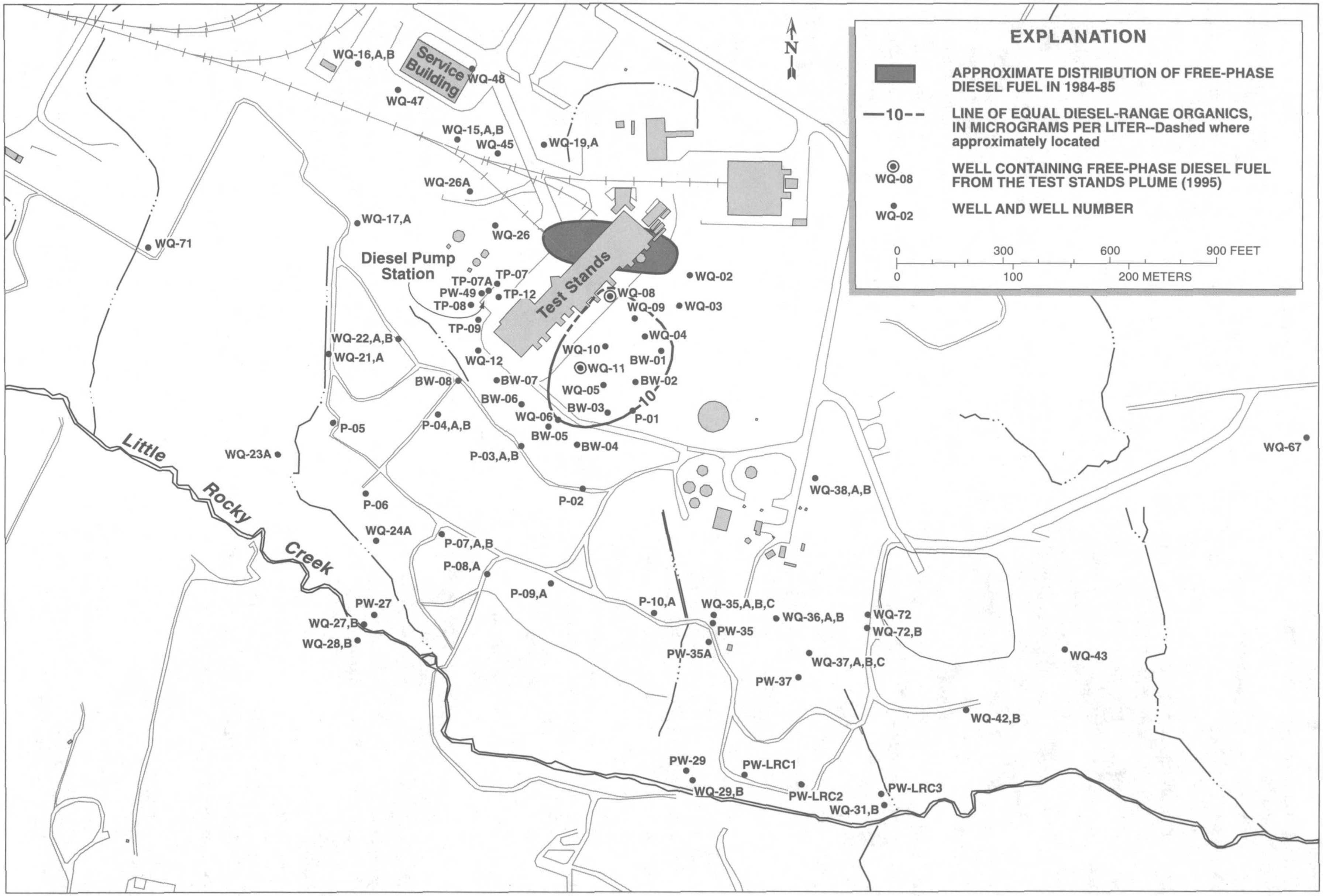


Figure 5. Location of wells containing free-phase diesel fuel from the Test Stands contamination in October 1995, distribution of diesel-range organics in ground water in October 1995 (modified from Kubal-Furr & Associates, 1996), and approximate distribution of free-phase diesel fuel in 1984-85 (Law Engineering and Testing Company, 1984a, 1984b, 1985) at a gas-turbine manufacturing facility, Greenville, S.C.

Despite the differences in data types over time, the data are sufficient to show that contaminants from the initial spill migrated southeastward beneath the Test Stands Building (fig. 5). Free-phase fuel was detected in well WQ-11 in December 1989 and in well WQ-08 in August 1991 (fig. 5). The absence of free-phase fuel in those wells during prior sampling events indicates lateral transport of the fuel (Sirrine Environmental Consultants, 1991b). The transport of fuel from the spill area to well WQ-11 implies that it is unreasonable to expect free-phase-fuel transport to have stopped at well WQ-11. However, the lack of detected free-phase fuel at wells BW-03, BW-04, BW-05, WQ-05, and WQ-06 indicate that fuel transport has not extended to those areas.

The transport of dissolved petroleum hydrocarbons differs from the transport of free-phase fuel partly because petroleum hydrocarbons are more readily biodegraded in dissolved form. For example, benzene concentrations did not substantially change at wells WQ-05, WQ-06, or WQ-10 between 1991 and 1997 (table 2), implying that the advective transport rate of benzene into those areas was balanced by the benzene removal rate. Factors controlling the benzene removal rate potentially include the rates of biodegradation and adsorption relative to the rate of advective transport. Slight increases in benzene concentrations observed at wells BW-03, BW-04, and WQ-04 between 1991 and 1997, imply that the contamination was migrating toward these wells.

No ground-water chemistry data are available delineating the vertical extent of petroleum hydrocarbon contamination immediately southeast of the Test Stands; however, the probable vertical extent can be estimated using a variety of factors. The petroleum hydrocarbon of greatest interest to this investigation is benzene. Because benzene is less dense than water, there is no tendency for the contamination to sink into the aquifer by density-driven movement. The larger horizontal hydraulic gradient at the Test Stands, compared to the vertical hydraulic gradient, also suggests that there is little tendency at that location for benzene to be transported significantly deeper into the aquifer by advective transport. Moreover, the upward hydraulic gradient from the transition zone to the saprolite in areas downgradient from well P-01 further limits downward movement of benzene. Therefore, it is highly probable that benzene in the Test Stands contamination is most concentrated in the saprolite.

Biodegradation Potential

Water from most wells sampled on the eastern side of the Test Stands (fig. 1) in October 1995 contained low concentrations of DO, ranging from 0.44 milligrams per liter (mg/L) in well WQ-09 to 0.98 mg/L in well P-01 (table 3). The lowest DO concentrations on the eastern side of the Test Stands were from wells WQ-05, -06, and -09, which also contained the highest concentrations of DRO's among the sampled wells (table 4). The low concentrations of DO is the result of oxygen depletion by aerobic microbial degradation of petroleum hydrocarbons.

It is probable that the petroleum hydrocarbon biodegradation near the main body of contamination occurs under anaerobic conditions. The presence of sulfate at 2.64 mg/L in ground water from the well with the highest concentration of DRO's (well WQ-05) and at only 0.75 mg/L at well WQ-09, closer to the spill area, implies that sulfate reduction is active in nearby anaerobic zones. Concentrations of sulfate above 2 mg/L have been observed to support sulfate reduction in other petroleum-hydrocarbon contaminated ground water (Vroblesky, Bradley, and Chapelle, 1996). The presence of methane in WQ-03, WQ-05, WQ-06, WQ-09, and P-02 in the vicinity of the Test Stands supports the hypotheses that anaerobic conditions are present in some parts of the contamination and that methanogenesis is an active TEAP in contaminated areas upgradient from those wells.

The wells in the vicinity of the Test Stands contamination containing the highest concentrations of DRO's also contained the highest concentrations of DIC. Because one source of DIC is from dissolved carbon dioxide production during petroleum-hydrocarbon biodegradation, the relatively high DIC concentrations in wells with high concentrations of DRO's support the hypothesis that the diesel fuel is being microbially degraded.

Evidence that selected components of diesel-fuel contamination are actively biodegraded in aquifer sediments from the facility also is shown by the laboratory data (table 5). The data indicate that native microbial populations are capable of degrading benzene and decane under aerobic conditions.

Currently, insufficient data are available to quantify the biodegradation potential of naphthalene in the study area. In general, naphthalene appears to be recalcitrant or to degrade more slowly than benzene in anaerobic contaminated sediment (Milelcic and Luthy, 1988; Landmeyer and others, 1998), apparently because the slow naphthalene desorption rate limits the amount of bioavailable naphthalene (Volkering and others, 1993; Ghoshal and others, 1996). Although aerobic naphthalene and benzene biodegradation rates have been reported to be similar (Landmeyer and others, 1998), the observations of limited naphthalene biodegradation implies the possibility that naphthalene could be more persistent than benzene at this site.

Table 2. Concentrations of benzene in ground water at selected wells from March 1991 to August 1997 at a gas-turbine manufacturing facility, Greenville, S.C. (data from Robert DeGaetano, Kubal-Furr and Associates, written commun., 1997; and W.Thomas Hyde, General Electric Co., written commun., 1997).

[μg/L, micrograms per liter; ---, data not collected; <, less than]

Well number	Concentration of benzene in μg/L												
	Mar. 1991	Jun. 1991	Aug. 1991	Sep. 1991	Dec. 1991	Nov. 1992	Oct. 1994	Jan. 1996	Oct. 1996	Feb. 1997	April 1997	May 1997	Aug. 1997
BW-01	<1	<1	---	---	---	---	---	---	---	---	---	---	---
BW-02	<1	<1	---	---	---	---	---	---	---	---	---	---	---
BW-03	1.3	<1	---	---	---	---	---	---	---	12.2	---	---	14.6
BW-04	1.1	2.1	---	---	---	---	---	---	---	14	---	---	8.7
P-01	<1	<1	---	<5	<1	<5	---	---	<2	---	---	---	---
P-02	<1	<1	---	<5	<1	---	---	---	<2	<5	---	<2	<5
WQ-02	<1	<1	---	<1	---	---	---	<5	---	---	---	---	---
WQ-03	<1	<1	<1	---	<1	<5	---	---	---	---	---	---	---
WQ-04	9.7	14	15	---	20	---	---	---	---	49.4	---	---	29
WQ-05	19	15	---	15	11	---	---	---	---	13.1	14.9	---	16.1
WQ-06	1	<1	---	<5	3.5	---	3.7	---	---	5.2	<5	---	<5
WQ-08	43	22	Free-phase fuel	---	---	---	---	---	---	---	---	---	---
WQ-09	17	14	13	---	11	---	---	---	---	---	---	---	---
WQ-10	7.8	23	16	---	16	---	---	17.6	15	---	16.8	---	---

Table 3. Inorganic constituents, field parameters, and methane in ground water at a gas-turbine manufacturing facility, Greenville, S.C., October 1995.

[DO, dissolved oxygen; Temp, water temperature; Fe(II), ferrous iron; H_2, hydrogen, Cl, chloride; Br, bromide; NO_3, nitrate; SO_4, sulfate; Na, sodium; K, potassium; Mg, magnesium; Ca, calcium; DIC, dissolved inorganic carbon; CH_4, methane; mg/L, milligrams per liter; s.u., standard units; °C, degrees Celsius; nM, nanomoles per liter; µM, micromoles per liter; ---, information not available; <, less than; (R), replicate sample; ammonium, nitrite, and phosphate concentrations were less than the detection limit of 0.02 mg/L]

Site number	DO (mg/L)	pH (s.u.)	Temp (°C)	Fe(II) (mg/L)	H_2 (nM)	Cl (mg/L)	Br (mg/L)	NO_3 (mg/L)	SO_4 (mg/L)	Na (mg/L)	K (mg/L)	Mg (mg/L)	Ca (mg/L)	DIC (mg/L)	CH_4 (µM)
P-01	0.98	4.94	17.3	---	---	3.72	0.05	2.38	0.70	2.27	0.56	0.56	0.07	116.3	<1.0
P-02	6.34	5.38	17.2	0.8	---	3.59	.07	4.20	1.11	2.08	.45	.74	.06	124.6	40.1
P-04	7.20	5.31	18.2	<.1	---	2.05	<.02	.80	.80	.98	.06	.15	.20	44.7	<1.0
P-04A	4.00	5.85	17.3	---	---	.97	<.02	.23	9.71	2.31	1.25	1.62	2.60	32.7	<1.0
P-04B	.25	6.44	18.3	---	---	.98	<.02	<.01	10.43	5.44	1.37	1.68	4.05	34.1	<1.0
P-09	3.60	5.66	16.9	---	---	2.38	.08	.90	.71	1.49	.84	.41	2.38	83.1	<1.0
TP-07	2.50	5.13	23.5	<.1	---	17.79	.07	.40	1.10	10.97	.19	.68	.08	175.4	36.6
TP-07A	8.50	5.69	19.6	<.1	---	.94	.03	.07	1.53	.38	.81	.35	.14	31.1	<1.0
TP-07A (R)	---	---	---	---	---	.87	<.02	.06	1.47	.38	.80	.34	.19	29.9	<1.0
WQ-03	2.56	5.15	16.9	---	---	2.10	.14	.10	1.34	1.60	.27	.61	.14	162.4	61.3
WQ-05	.53	4.82	18.2	.6	---	5.62	.21	.09	2.64	1.57	.45	1.75	.32	243.5	220.5
WQ-06	.73	4.82	20.5	.4	---	4.18	<.02	.27	1.03	1.34	.35	1.94	<.04	406.2	71.9
WQ-09	.44	5.14	17.1	---	---	2.20	.12	<.01	.75	1.38	1.02	1.35	.43	212.4	141.6
WQ-15	4.86	5.82	21.0	<.1	---	2.71	<.02	4.53	2.85	3.72	.04	.04	.38	29.9	<1.0
WQ-15A	.15	6.18	18.6	3.4	---	1.10	.03	.04	29.19	8.23	8.13	7.29	3.39	42.0	<1.0
WQ-15B	.04	6.04	19.1	3.6	2.4	.95	<.02	<.01	17.81	10.33	4.66	2.74	4.74	36.9	<1.0
WQ-24A	1.52	5.98	18.9	---	---	1.22	<.02	<.01	11.08	4.52	1.81	1.62	5.06	30.0	<1.0
WQ-26A	4.98	6.48	19.1	---	---	1.07	.06	.24	12.25	9.47	6.08	3.67	10.39	29.1	<1.0
WQ-27	6.00	5.36	21.0	---	---	1.27	<.02	<.01	5.64	2.08	1.73	.25	6.67	93.4	<1.0
WQ-27B	1.29	6.99	17.6	---	---	1.35	<.02	.18	6.27	9.39	3.52	4.23	24.15	48.5	<1.0
WQ-28	9.20	5.52	18.6	---	---	---	---	---	---	2.10	.27	.54	.62	38.4	<1.0

Table 4. Organic constituents in ground water at a gas-turbine manufacturing facility, Greenville, S.C., October 1995 (Kubal-Furr, 1996).

[1,1,1-TCA, 1,1,1-trichloroethane; 1,1,2-TCA, 1,1,2-trichloroethane; 1,1-DCA, 1,1-dichloroethane; 1,1-DCE, 1,1-dichloroethene; *c* 1,2-DCE, *cis* 1,2-dichloroethene; *t* 1,2-DCE, *trans* 1,2-dichloroethene; MC, methylene chloride; PCE, tetrachloroethene; TCE, trichloroethene; VC, vinyl chloride; CF, chloroform; Benz, benzene; Tol, toluene; E-benz, ethylbenzene; Xyl, total xylenes; DRO, diesel-range organics; µg/L, micrograms per liter; —, information not available; <, less than; J, result is an estimated quantity; /B, result is below the sample quantitation limit and above or equal to the instrument detection limit; (R), replicate sample]

Site number	1,1,1-TCA µg/L	1,1,2-TCA µg/L	1,1-DCA µg/L	1,1-DCE µg/L	c 1,2-DCE µg/L	t 1,2-DCE µg/L	MC µg/L	PCE µg/L	TCE µg/L	VC µg/L	CF µg/L	Benz µg/L	Tol µg/L	E-benz µg/L	Xyl µg/L	DRO µg/L
P-01	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	85J
P-02	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	<10.0
P-04	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	2.3 J/B	23.5	<5.0	<10.0	3.9 J/B	<5.0	<5.0	<5.0	<5.0	<10.0
P-04A	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	1,230	9.2	<10.0	<5.0	<5.0	<5.0	<5.0	<5.0	—
P-04B	<5.0	<5.0	<5.0	<5.0	1.8 J	<5.0	<5.0	249	21.5	<10.0	<5.0	<5.0	<5.0	<5.0	<5.0	—
P-07	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	<10.0
P-09	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	<10.0
TP-07	2.0 J	12.3	36.7	107	35,200	162	1,450	2,580	626	1,630	6.6	9.2	14.1	5.6	155	—
TP-07 (R)	2.0 J	12.1	35.6	99.8	46,300	155	1,440	1,820	642	1,700	6.3	8.1	12.4	4.8	152	—
TP-07A	<5.0	<5.0	<5.0	<5.0	14.0	<5.0	2.2 J/B	402	3.4 J	<10.0	4.5 J/B	<5.0	<5.0	<5.0	<5.0	—
WQ-03	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	39J
WQ-05	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	14.9
WQ-06	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	10.0
WQ-09	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	10.7
WQ-09 (R)	—	—	—	—	—	—	—	—	—	—	—	—	—	—	—	12.9
WQ-15	<5.0	<5.0	<5.0	<5.0	40.6	<5.0	3.6 J/B	790	2.0 J	<10.0	4.5 J/B	<5.0	1.9 J	4.4J	2.8J	—
WQ-15A	<5.0	<5.0	<5.0	<5.0	11.4	<5.0	1.0 J/B	231	14.9	<10.0	4.1 J/B	<5.0	<5.0	<5.0	<5.0	—
WQ-15B	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<10.0	3.6 J/B	<5.0	<5.0	<5.0	<5.0	—
WQ-24A	<5.0	<5.0	<5.0	<5.0	7.5	<5.0	1.5 J/B	172	11.2	<10.0	<5.0	<5.0	<5.0	<5.0	<5.0	—
WQ-26A	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<10.0	<5.0	<5.0	<5.0	<5.0	<5.0	—
WQ-26A (R)	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<10.0	<5.0	<5.0	<5.0	<5.0	<5.0	—
WQ-27	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	7.6	<5.0	<10.0	<5.0	<5.0	<5.0	<5.0	<5.0	—
WQ-27B	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	3.4 J	<5.0	<10.0	<5.0	<5.0	<5.0	<5.0	<5.0	—
WQ-28	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<5.0	<10.0	<5.0	<5.0	<5.0	<5.0	<5.0	—
WQ-45	3.3 J	54.2	3.2 J	117	79,000	76.6	<5.0	5,500	565	478	2.2 J	10.6	76.9	85	272	—

Table 5. Laboratory-derived biodegradation rates, in percentage degraded per day (plus or minus the standard deviation), of benzene and decane using sediment microcosms from a gas-turbine manufacturing facility, Greenville, S.C.

[±, plus or minus]

Approximate location of sediment collection site	Percentage of initial concentration degraded per day	
	Benzene	Decane
Background	16.3 ± 2.0	4.37 ± 3.37
WQ-11	1.25 ± 0.15	0.07 ± 0.05
WQ-10 (auger cuttings)	1.30 ± 0.6	7.20 ± 5.1

Diesel Pump Station Contamination

In addition to the Test Stands fuel-line leak, petroleum-hydrocarbon and CVOC contamination is present in ground water at the Diesel Pump Station (fig. 1). The contamination was discovered in 1984 during investigation of the Test Stands contamination (Law Engineering Testing Company, 1984a).

Distribution

Ground-water contamination associated with the Diesel Pump Station originates near the southwestern edge of the Test Stands building (fig. 6). The contamination consists of a mixture of CVOC's and DRO's from diesel fuel (Davis and Floyd, Inc., 1994). Contamination by CVOC's from the Diesel Pump Station contamination and the Service Building contamination probably commingle at depth in the saprolite/transition-zone aquifer (fig. 7).

Petroleum hydrocarbons from the Diesel Pump Station contamination and the Test Stands contamination also probably commingle in downgradient areas. The direction of ground-water flow in the saprolite/transition-zone aquifer is southeastward from the Diesel Pump Station (fig. 3). This path of potential contaminant transport approximately intersects the southern part of the Test Stands contamination (fig. 5). Thus, the petroleum hydrocarbon contamination in the southern end of the Test Stands contamination may be derived partly from the Diesel Pump Station.

The primary CVOC's present in the saprolite/transition-zone aquifer at the Diesel Pump Station in October 1995 were *cis* 1,2-DCE (35,200 µg/L) and PCE (2,580 µg/L) (table 4, well TP-07). When highly concentrated, these compounds have a tendency to migrate downward in aquifers because they are denser than water. The presence of PCE in the underlying transition zone at well TP-07A (table 4) implies that the contamination has migrated vertically downward into the transition zone (fig. 7). The presence of 249 µg/L of PCE at well P-04B, downgradient from the Diesel Pump Station, shows that CVOC contamination also has migrated downward into the fractured bedrock (table 4, fig. 7).

The CVOC contamination from the Diesel Pump Station is approximately downgradient from sources of CVOC contamination near the Test Stands. Thus, the contamination from both sources probably commingles in the saprolite/transition-zone aquifer (fig. 7). The areal distribution of the CVOC contamination implies that the contaminant transport direction is approximately southward in the deeper part of the flow system (fig. 6).

Biodegradation Potential

The presence of CVOC's and BTEX compounds in ground-water contamination can accelerate the degradation of each other (Sewell and Gibson, 1991). The reason for the accelerated degradation is that higher chlorinated CVOC's typically are degraded by reduction, and BTEX compounds typically are degraded by oxidation. Because oxidation reactions are coupled to reducing reactions, the commingling of oxidizable and reducible contaminants provides both the electron donor and the electron acceptor necessary to sustain the redox reactions.

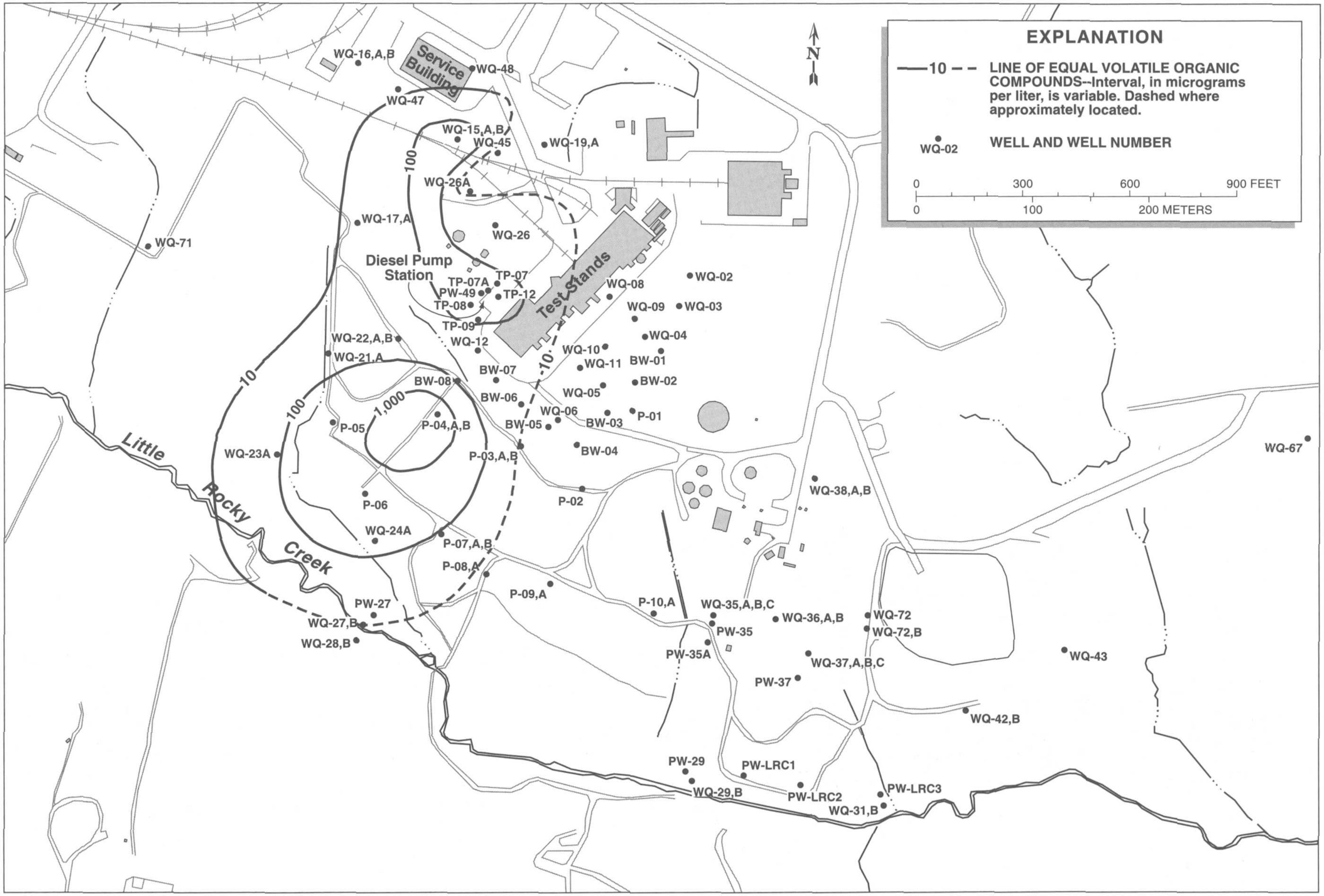


Figure 6. Distribution of volatile organic compounds in ground water in the transition zone between saprolite and bedrock at a gas-turbine manufacturing facility, Greenville, S.C., October 1995 (Kubal-Furr & Associates, 1996).

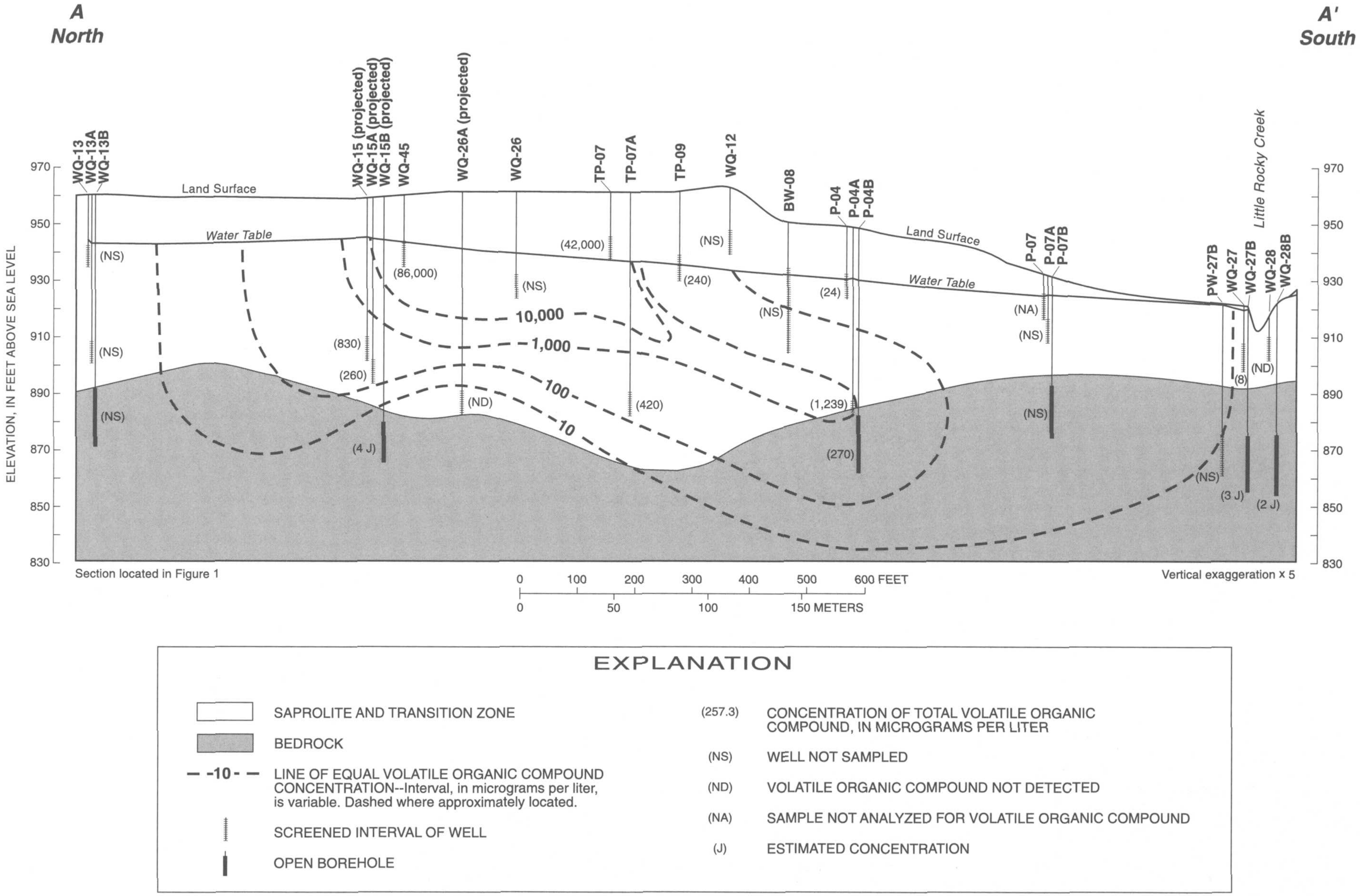


Figure 7. Distribution of volatile organic compounds in ground water along hydrogeologic section *A-A'* at a gas-turbine manufacturing facility, Greenville, S.C., October 1995 (modified from Kubal-Furr & Associates, 1996).

Evidence for such extensive degradation at the facility is the relatively high concentration of DIC at well TP-07 (175.4 mg/L) as well as the presence of several dechlorination products, such as 1,1-dichloroethene, *cis* 1,2-dichloroethene, *trans* 1,2-dichloroethene, methylene chloride, and VC. *Cis* 1,2-dichloroethene is present at a concentration of 35,200 μg/L. The ground water at well TP-07 is aerobic (2.5 mg/L of DO). In other investigations, VC has been shown to degrade rapidly under aerobic conditions (Hartman and others, 1985; Davis and Carpenter, 1990; Phelps and others, 1991). Thus, the presence of VC (1,630 μg/L) in aerobic ground water at well TP-07 suggests that extensive dechlorination of higher chlorinated compounds maintains elevated VC concentrations near the Diesel Pump Station. The dechlorination may take place in local anaerobic zones associated with petroleum-hydrocarbon contamination.

Further evidence of dechlorination in the aquifer at well TP-07 is suggested by the chloride concentrations. In October 1995, the ground water at well TP-07 contained 17.79 mg/L of chloride, which was substantially higher than the typical concentration of less than 6 mg/L at most wells. A possible source of chloride is dechlorination from higher to lower chlorinated aliphatic compounds in the aquifer.

Examination of water chemistry from wells P-04A (transition zone) and well P-04B (fractured bedrock) can provide information on the downgradient potential for PCE transformations (fig. 1). The ground water at wells P-04A and P-04B contained 1,230 μg/L and 249 μg/L of PCE, respectively, in October 1995 (table 4). The concentrations of chloride, 0.97 and 0.98 mg/L, respectively, in those zones, however, were lower than at most other wells (table 3), implying that dechlorination reactions are not significant in this area. Moreover, the presence of 4 mg/L of DO in ground water at well P-04A and 7.2 mg/L of DO in the overlying saprolite at well P-04 (table 3) suggests that aerobic conditions will inhibit further dechlorination of PCE along the path of flow. The same aerobic conditions, however, probably will prevent extensive migration of VC.

Service Building Contamination

The ground-water contaminants associated with the Service Building area are primarily PCE and DCE, with lesser concentrations of TCE, as well as petroleum hydrocarbons. Ground-water contamination by CVOC's downgradient from the Service Building appears to have originated from several areas. In addition to the Diesel Pump Station, previously discussed, these areas include those near the Service Building, well WQ-45, and possibly near the National Pollutant Discharge Elimination System (NPDES) outfall-002 (SEC Donohue, 1992a) (fig. 1). The Service Building was used to store waste cuttings, grinding oil, and drums of new and used chlorinated solvents prior to 1976 (SEC Donohue, 1992a; Kubal-Furr and Associates, 1996).

Distribution

The ground-water contamination from the Service Building is moving south to southwestward toward Little Rocky Creek (fig. 6). Concentrations of PCE up to 30,000 μg/L in ground water (well WQ-45) were reported in 1992 (Davis and Floyd, Inc., 1994). Volatile organic compounds have been detected adjacent to Little Rocky Creek at wells WQ-27 and WQ-27B (Kubal-Furr and Associates, 1996) (fig. 6). The presence of VOC's in wells adjacent to Little Rocky Creek indicates that ground-water contaminants from the Service Building, the Diesel Pump Station, or a combination of contaminants from the two source areas has reached the creek and probably discharges to the creek.

The presence of VOC contamination in deeper parts of the aquifer near the source areas (fig. 7) implies that chlorinated compounds from the Service Building contamination migrated vertically downward under a density gradient through the saprolite into the transition zone and, probably, into the fractured bedrock (fig. 7). The presence of PCE (11.7 μg/L) in ground water collected from transition-zone well WQ-17A (Kubal-Furr and Associates, 1996), screened 96 to 106 ft below land surface, indicates that the Service Building contamination has been transported laterally at depth.

Biodegradation Potential

As with the CVOC contamination from the Diesel Pump Station, the CVOC contamination from the Service Building shows a high potential for contaminant degradation in parts of the source area, but little potential for degradation for those CVOC's transported away from the source area. One such area showing a high potential for contaminant degradation is at well WQ-45 (fig. 1). The ground water at well WQ-45 contained the highest concentrations of PCE (5,500 μg/L) among the wells sampled in October 1995 for this investigation. Petroleum hydrocarbons also were present in the aquifer at well WQ-45, as evidenced by detection of BTEX compounds (table 4). It is clear from the presence of 79,000 µg/L of *cis* 1,2-dichloroethene and 478 µg/L of VC that a substantial amount of dechlorination is taking place in the aquifer near the well. Such extensive dechlorination is consistent with accelerated degradation caused by commingling of CVOC's and petroleum hydrocarbons (Sewell and Gibson, 1991).

At well WQ-26, located 200 ft approximately downgradient from well WQ-45, however, the data imply that PCE has been transported downgradient. Although well WQ-26 was not sampled during October 1995, PCE commonly was detected (13 sampling dates between August 1992 and January 1996) at concentrations exceeding 1,000 µg/L (Robert DeGaetano, Kubal-Furr and Associates, written commun., 1997). Aerobic conditions in ground water at wells P-04 and P-04A imply that PCE biodegradation is limited in downgradient areas.

SIMULATED GROUND-WATER FLOW AND BENZENE TRANSPORT IN THE SAPROLITE/TRANSITION-ZONE AQUIFER

A ground-water flow model and a solute-transport model were used to simulate the potential for natural attenuation of benzene in the Test Stands contamination area. The flow simulations provided a description of hydraulic potential in the combined saprolite/transition-zone aquifer. A solute-transport model simulated potential benzene-attenuation mechanisms, such as bioremediation. Coupling the flow and solute-transport simulations allowed estimates to be made as to whether the benzene-attenuation mechanisms were extensive enough, relative to ground-water flow, to prevent contaminant transport to offsite areas or to Little Rocky Creek. Ground-water flow was simulated using the BOSS International (1996) GMS version of the U.S. Geological Survey modular 3-D flow model (MODFLOW) (McDonald and Harbaugh, 1988).

Transport of the ground-water contamination was simulated using BOSS International (1996) GMS version of MT3D, a three-dimensional finite-difference code that can simulate advection, dispersion, sink/source mixing, adsorption, and chemical reactions (Zheng, 1990). The code uses a modular structure similar to the structure used by MODFLOW and can be used in conjunction with MODFLOW in a two-step flow and transport simulation. Heads and cell-to-cell flux terms are computed by MODFLOW during simulation of flow and are used by MT3D as the flow field to compute transport. A mathematical description of the code is found in Zheng (1990).

Ground-Water Flow Model

The USGS modular ground-water flow model, MODFLOW, solves the partial-differential ground-water flow equation using the block-centered finite-difference method (McDonald and Harbaugh, 1988). The result is a system of simultaneous linear equations iteratively solved by the preconditioned conjugate gradient solver (PCG2). The steady-state solution yields values of head and flow at specific points in space.

The modeled area consisted of one layer having 133 rows oriented east to west and 127 columns oriented north to south (fig. 8). Linear cell dimensions over most of the modeled area were 25 ft by 25 ft, and the column width expanded to 50 ft near the east and west boundaries. Spatial variation in cell size allowed finer discretization in the area of interest and coarser discretization outside of the area of interest where less information was available. The dimensions of the modeled area were approximately 0.64 mi by 0.77 mi. The simulated aquifer properties included hydraulic conductivity and ground-water levels. Simulated stresses included recharge from rainfall and discharge by baseflow to streams and to pumped wells. The flow solution was run under steady-state conditions because the stresses (recharge and pumpage) did not significantly vary within individual simulation periods.

Boundary Conditions and Simulated Stresses

The model boundaries were areal sources, sinks, or barriers for ground-water flow in the saprolite/transition-zone aquifer. The source of water was recharge infiltration into the upper model boundary. The upper model boundary was defined as the water table and was simulated as variable-head cells (free to fluctuate). The lower model boundary was simulated as a no-flow boundary at the approximate interface of the transition-zone and the underlying fractured bedrock, as defined by SEC Donohue (1992a).

Figure 8. Finite-difference grid and model boundaries at a gas-turbine manufacturing facility, Greenville, S.C.

At the southern end of the model, ground-water discharge to Little Rocky Creek constituted a hydraulic sink in the saprolite/transition-zone aquifer (fig. 8). The creek was simulated as a line of river cells. Values used to represent streambed conductance were 130, 182, 250, and 260 ft^2/d; the variations dominantly being the result of incorporating differences in model-cell size with a uniform hydraulic conductivity of 1.3 ft/d. The simulated hydraulic conductivity value was slightly less than the average hydraulic conductivity measured for the study area during previous investigations. The creek-bed elevation was estimated from topographic maps (5-ft contour interval), and the stage of the creek was incorporated as an addition to the stream base.

A general-head boundary approximately 50 to 100 ft south of Little Rocky Creek constitutes the southernmost boundary and provided a flux of ground water into the model grid that discharged into the southern side of the creek. The conductance used to simulate the southern general-head boundary was 50 ft^2/d. This value was chosen to approximate the drain conductance. Test simulations showed that doubling and halving the conductances along this boundary resulted in increasing discharge by 13 percent and decreasing it by 11 percent, respectively, but had little effect on model heads [the root mean squared error (RMSE) changed by a maximum of 0.03 ft]. The resulting simulated discharges were within the range of uncertainty associated with the estimated discharge. Boundary heads were set approximately to average heads where data existed and to a reflection across Little Rocky Creek of simulated or field-derived average-annual heads north of the creek where data were lacking.

The northwestern model boundary was oriented along a ground-water divide (figs. 3 and 8) identified from several synoptic water-level measurements made during previous investigations (Davis and Floyd, 1994; Kubal-Furr and Associates, 1996). The northwestern edge of the model, therefore, was simulated as a no-flow boundary.

At most areas along the eastern edge of the model, ground water discharges to a drainage ditch that is a tributary to Little Rocky Creek. In those areas, general-head cells with a conductance of 50 ft^2/d were used to simulate the boundary. Test simulations showed that doubling and halving the conductances along this boundary resulted in increasing discharge by 0.5 percent and decreasing it by 0.4 percent, respectively, and had little effect on model heads [the root mean squared error (RMSE) changed by a maximum of 0.05 ft]. The head values used to simulate this boundary were a uniform addition of one foot (estimated water depth) to the estimated stream base derived from a topographic map. At areas along the northeastern boundary where ground water does not discharge to the ditch, the model edge was oriented along an approximate ground-water flowline (Rust Environment and Infrastructure, 1995a; Kubal-Furr and Associates, 1996) and simulated as a no-flow boundary (fig. 8). At various locations on the interior of the modeled area, tributaries to Little Rocky Creek were simulated as drain cells with a hydraulic conductance of 60 ft^2/d, calculated in a similar manner to that used for streambed conductance in Little Rocky Creek.

Simulated stresses in this investigation include recharge as a source of water and pumping from wells as a sink for water. Simulated recharge to this model was based on initial calibration runs and compared to measured recharge rates from saprolite in North Carolina (Daniel, 1996) to ensure consistency. A simulated recharge value of 5.4 in./yr during non-pumping conditions produced a reasonable amount of simulated discharge to Little Rocky Creek (slightly less than the measured increase in creek flow from baseflow and runoff through the modeled area). Therefore, this value was used to represent recharge during non-pumping conditions. Because the average annual rainfall was slightly higher during the pumping period than the non-pumping period (National Oceanic and Atmospheric Administration, 1996), a value of 6 in./yr was used to simulate recharge under pumping conditions.

Simulated average-annual pumpage used in this investigation was estimated from site records. The simulated pumping rates were 955 ft^3/d from well PW-LRC2; 1,630 ft^3/d from well PW-LRC3; and 270 ft^3/d from well PW-35A. These values were within one standard deviation of recorded values, a difference that was insignificant to the flow solution (as will be shown in the section on sensitivity analysis).

Calibration Procedure, Criteria, and Results

Flow-model calibration was accomplished by matching simulated water levels to field-derived average-annual water levels, by maintaining simulated discharge to Little Rocky Creek at a value less than the measured increase in creek flow through the modeled area, and by approximately matching simulated hydraulic conductivity to values obtained by slug tests.

The model was calibrated to both non-pumping and pumping conditions. The water table under non-pumping conditions was simulated using field-derived average-annual water levels for April 1991 through April 1994 as the calibration points (fig. 9). The pumping period was simulated using measured average-annual water levels for June 1994 through July 1997 as the calibration points (fig. 10).

Initial modeling runs indicated that although simulated hydraulic conductivity exhibited a significant influence on the amount of simulated ground-water discharge to Little Rocky Creek, the dominant control was the amount of recharge added to the model. Thus, the initial calibrations included adjusting the recharge to produce an amount of discharge to Little Rocky Creek consistent with amount of flow increase measured in the creek during a previous investigation (SEC Donohue Inc., 1992a). The reported average stream-flow increase across the reach of Little Rocky Creek approximating the reach simulated in this investigation (11,000 ft^3/d) for November 1991 included runoff and was considered to constitute an upper-end constraining value to the simulated ground-water discharge in the creek. The simulated discharge to Little Rocky Creek was about 8,800 ft^3/d. Approximately 60 percent of the simulated discharge water was derived from the active portion of the modeled area north of Little Rocky Creek, the remaining discharge was derived from the model area south of Little Rocky Creek. Differences between November 1991 water levels and field-derived average-annual water levels were small relative to the range of measured values at most wells (table 6), implying that the simulated discharge is reasonable for the non-pumping simulation.

Once simulated recharge was adjusted to produce an acceptable amount of discharge to Little Rocky Creek, simulated hydraulic-conductivity values were adjusted to produce a close match between simulated and field-derived average-annual non-pumping water levels (fig. 11). A hydraulic conductivity of 2 ft/d was used for simulations in the immediate vicinity of the diesel-fuel contamination east of the Test Stands. This value was derived using a best-fit approach during model calibration after assuming an initial hydraulic conductivity value approximately the same as the average value (2.6 ft/d) determined during a previous investigation (SEC Donohue, 1992a). The final simulated hydraulic conductivity of 2 ft/d in the area immediately east of the Test Stands was obtained by using a best-fit approach during model calibrations.

Although no aquifer-test data were available for the northern part of the modeled area, simulated hydraulic conductivity adjustments made during model calibration suggest that a hydraulic conductivity of 2 ft/d also is appropriate for this area. Reliable aquifer test data were not found for the immediate vicinity of the Test Stands contamination. Much of this area was simulated using 0.3 ft/d and 0.5 ft/d because model simulations during calibration suggested that a value lower than 2 ft/d was appropriate (fig. 11).

In the southeastern and much of the eastern part of the modeled area, a hydraulic conductivity value of 0.5 ft/d (similar to the measured value from slug tests) was used to simulate the aquifer (fig. 11). Flow simulations during calibration implied that a slightly higher value of about 5 ft/d was appropriate for the southeastern part of the model near Little Rocky Creek. Thus, a hydraulic conductivity of 5 ft/d was used to simulate ground-water flow in the southeastern corner of the model and in the vicinity of well WQ-38, where the measured hydraulic conductivity was 7.7 ft/d (SEC Donohue, 1992a). The southern edge of the model near Little Rocky Creek was simulated using a hydraulic conductivity of 3 ft/d to approximate slug-test measurements at wells WQ-29 (2.9 ft/d) and WQ-34 (3.2 ft/d) (SEC Donohue, 1992a).

The zone of low ground-water levels south of the Test Stands (fig. 3) was interpreted as an area of significantly higher hydraulic conductivity than the surrounding values (fig. 11). As will be shown in the section on sensitivity analysis, doubling or halving the hydraulic conductivity in this zone had relatively little effect on the flow solution.

Adjustments to aquifer properties were made until a close match was achieved between field-derived average-annual non-pumping water levels at 43 wells and corresponding simulated heads for non-pumping conditions (table 7). The final average root mean squared error (RMSE) between simulated and field-derived average-annual conditions across the modeled area for the non-pumping simulation was 1.66 ft, or approximately 4 percent of the total head change across the modeled area. In the vicinity of the Test Stands contamination (wells P-01, WQ-02, WQ-03, WQ-04, WQ-05, WQ-06, WQ-09, and WQ-10) differences between simulated and field-derived average-annual water levels were less than 2 ft.

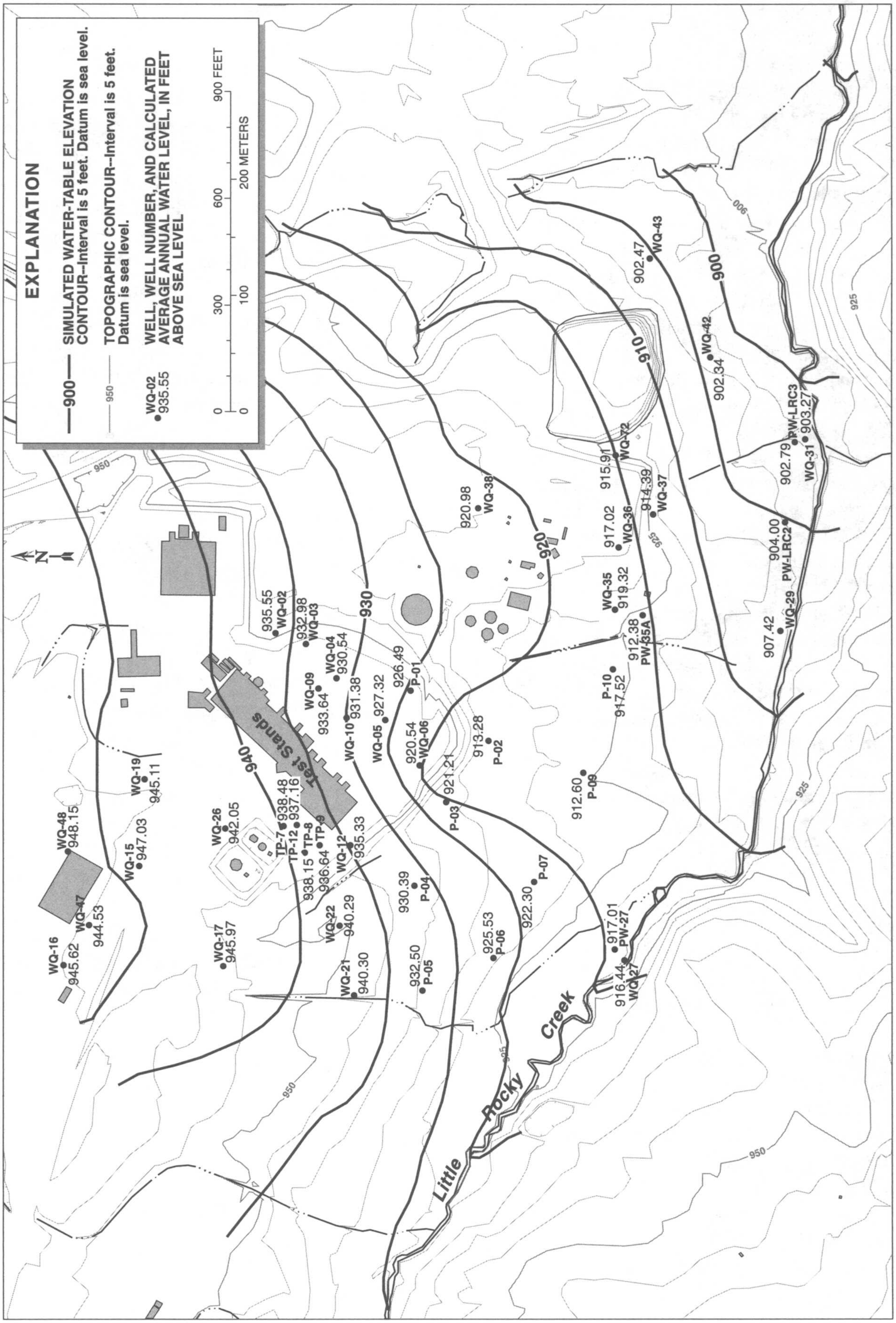


Figure 9. Simulated and field-derived average annual water-table elevations for non-pumping conditions, April 1991 through April 1994, at a gas-turbine manufacturing facility, Greenville, S.C.

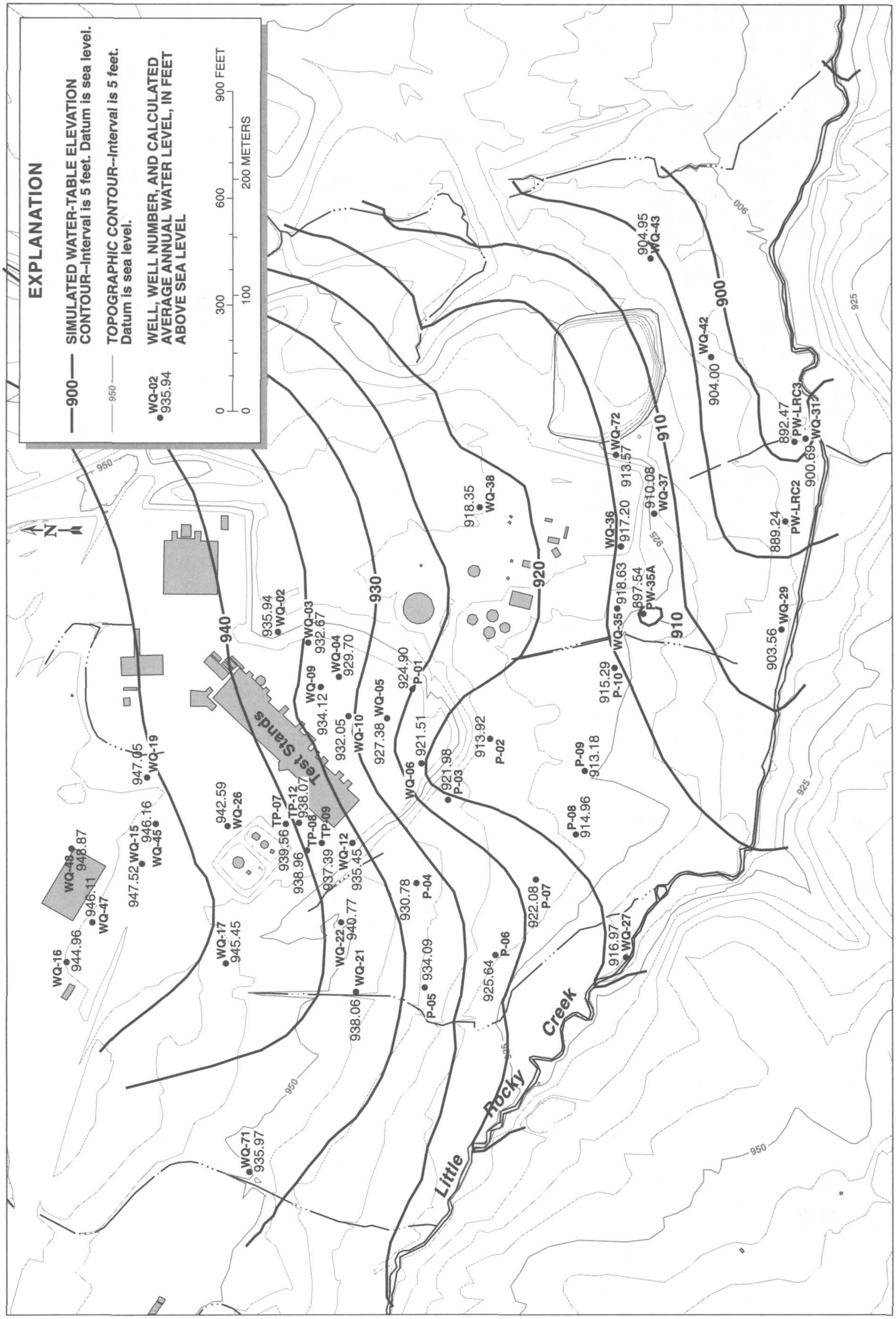


Figure 10. Simulated and field-derived average annual water-table elevations for pumping conditions, June 1994 through July 1997, at a gas-turbine manufacturing facility, Greenville, S.C.

Table 6. November 1991 and field-derived average-annual water levels during non-pumping conditions at a gas-turbine manufacturing facility, Greenville, S.C.

[ft, feet; sl, sea level]

Well number (fig. 1)	Water levels in November 1991 (ft relative to sl)	Field-derived average-annual water levels (ft relative to sl)	Field-derived average-annual heads minus November 1991 heads (ft)	Field-derived range of measured water levels (ft)
P-01	927.04	926.49	-0.55	6.92
P-02	912.88	913.28	.40	2.89
P-03	919.73	921.21	1.48	5.58
P-04	924.77	930.39	5.62	13.00
P-09	912.36	912.60	.24	2.30
P-10	917.49	917.52	.03	8.48
TP-08	936.56	938.15	1.59	7.83
TP-09	934.89	936.64	1.75	8.05
TP-12	935.73	937.16	1.43	7.60
WQ-02	934.88	935.55	.67	6.77
WQ-03	932.92	932.98	.06	7.75
WQ-04	930.79	930.54	-.25	7.96
WQ-05	927.49	927.32	-.17	8.58
WQ-06	920.32	920.54	.22	3.97
WQ-09	933.58	933.64	.06	7.98
WQ-10	929.39	931.38	1.99	7.41
WQ-29	906.97	907.42	.45	2.77
WQ-30	906.72	907.10	.38	2.23
WQ-31	903.04	903.27	.23	3.49
WQ-35	919.84	919.32	-.52	5.11
WQ-36	916.80	917.02	.22	6.31
WQ-37	913.75	914.39	.64	6.50
WQ-38	920.27	920.98	.71	5.12
WQ-42	900.98	902.34	1.36	8.10
WQ-43	900.14	902.47	2.33	8.04

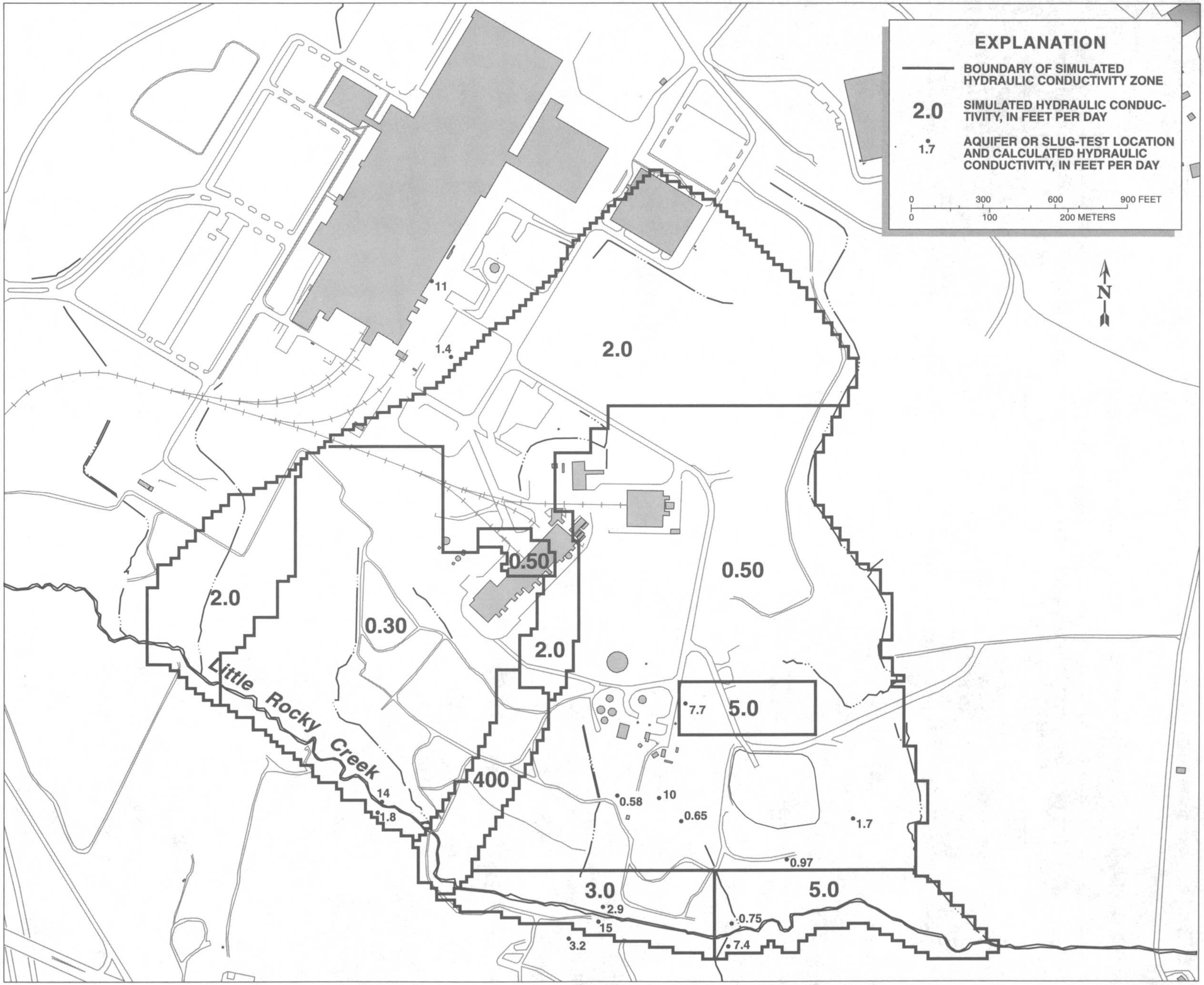


Figure 11. Data from hydraulic testing (SEC Donohue, 1992a) and simulated hydraulic conductivity at a gas-turbine manufacturing facility, Greenville, S.C.

Table 7. Field-derived average-annual and simulated water levels at a gas-turbine manufacturing facility, Greenville, S.C.

[ft, feet; sl, sea level; ---, data not available; *, data are from a pumping well]

	Non-pumping conditions: April 1991 through April 1994			Pumping conditions: June 1994 through July 1997		
Well number	**Field-derived average annual water level (ft relative to sl)**	**Simulated average annual water level (ft relative to sl)**	**Simulated minus field-derived average annual water level (ft)**	**Field-derived average annual water level (ft relative to sl)**	**Simulated average annual water level (ft relative to sl)**	**Simulated minus field-derived average annual water level (ft)**
P-01	926.49	924.80	-1.69	924.90	925.30	0.40
P-02	913.28	916.10	2.82	913.92	916.10	2.18
P-03	921.21	920.40	-.81	921.98	920.80	-1.18
P-04	930.39	931.20	.81	930.78	932.10	1.32
P-05	932.50	932.50	.00	934.09	933.20	-.89
P-06	925.53	926.70	1.17	925.64	927.10	1.46
P-07	922.30	922.50	.20	922.08	922.90	.82
P-08	---	917.60	---	914.96	917.70	2.74
P-09	912.60	915.90	3.30	913.18	916.00	2.82
P-10	917.52	916.80	-.72	915.29	915.20	-.09
PW-35A	912.38	915.40	3.02	897.54	906.40	8.86*
PW-LRC2	904.00	905.30	1.30	889.24	902.60	13.36*
PW-LRC3	902.79	902.20	-.59	892.47	895.70	3.23*
TP-07	938.48	939.80	1.32	939.56	941.00	1.44
TP-08	938.15	939.10	.95	938.96	940.30	1.34
TP-09	936.64	937.40	.76	937.39	938.60	1.21
TP-12	937.16	938.70	1.54	938.07	939.90	1.83
WQ-02	935.55	936.20	.65	935.94	937.00	1.06
WQ-03	932.98	934.30	1.32	932.67	935.10	2.43
WQ-04	930.54	931.10	.56	929.70	931.80	2.10
WQ-05	927.32	925.40	-1.92	927.38	925.90	-1.48
WQ-06	920.54	921.30	.76	921.51	921.60	.09
WQ-09	933.64	932.40	-1.24	934.12	933.20	-.92

Table 7. Field-derived average-annual and simulated water levels at a gas-turbine manufacturing facility, Greenville, S.C.--Continued

[ft, feet; sl, sea level; ---, data not available; *, data are from a pumping well]

	Non-pumping conditions: April 1991 through April 1994			Pumping conditions: June 1994 through July 1997		
Well number	**Field-derived average annual water level (ft relative to sl)**	**Simulated average annual water level (ft relative to sl)**	**Simulated minus field-derived average annual water level (ft)**	**Field-derived average annual water level (ft relative to sl)**	**Simulated average annual water level (ft relative to sl)**	**Simulated minus field-derived average annual water level (ft)**
WQ-10	931.38	930.00	-1.38	932.05	930.70	-1.35
WQ-12	935.33	934.50	-.83	935.45	935.50	.05
WQ-15	947.03	944.70	-2.33	947.52	946.10	-1.42
WQ-16	945.62	945.60	-.02	944.96	947.10	2.14
WQ-17	945.97	942.90	-3.07	945.45	944.30	-1.15
WQ-19	945.11	944.20	-.91	947.05	945.40	-1.65
WQ-21	940.30	937.00	-3.30	938.06	938.00	-.06
WQ-22	940.29	938.00	-2.29	940.77	939.20	-1.57
WQ-26	942.05	943.10	1.05	942.59	944.40	1.81
WQ-27	916.44	918.80	2.36	916.97	918.80	1.83
WQ-29	907.42	907.90	.48	903.56	907.70	4.14
WQ-31	903.27	901.90	-1.37	900.69	899.90	-.79
WQ-35	919.32	916.90	-2.42	918.63	913.80	-4.83
WQ-36	917.02	916.40	-.62	917.20	914.90	-2.30
WQ-37	914.39	913.30	-1.09	910.08	911.90	1.82
WQ-38	920.98	920.50	-.48	917.92	920.70	2.78
WQ-42	902.34	902.70	.36	904.00	902.00	-2.00
WQ-43	902.47	905.40	2.93	904.95	905.50	.55
WQ-47	944.53	945.70	1.17	946.11	947.10	0.99
WQ-48	948.15	945.70	-2.45	948.87	947.10	-1.77
WQ-69	---	947.90	---	946.39	949.10	2.71
WQ-71	---	935.60	---	935.97	936.30	.33
WQ-72	915.91	914.70	-1.21	913.57	914.10	.53

When the differences between simulated and field-derived average-annual water levels at most wells were less than about 3 ft and simulated hydraulic conductivity values approximated measured values, the calibrated model was run under pumping conditions. The steady-state pumping period represented June 1994 to July 1997. Ground-water removal by pumping was simulated at recovery wells southeastward from the Test Stands. Recharge was simulated as 6.0 in./yr, slightly higher than non-pumping recharge (5.4 in./yr) to account for increased water levels in some wells outside the influence of pumping wells and a slight increase in measured rainfall (National Oceanic and Atmospheric Administration, 1996).

The final average RMSE between simulated and field-derived average-annual water levels across the modeled area during pumping simulations was 2.64 ft, or less than 6 percent of the total head change across the modeled area. In all cases, simulated water levels at pumped wells were higher than field-derived average-annual water levels, as would be expected with a grid spacing of 25 ft. When pumped wells were excluded from the comparison, the average RMSE declined to 1.84 ft for pumping conditions, and in the vicinity of the Test Stands contamination, differences between simulated and field-derived average-annual water levels were 2.43 ft or less under pumping conditions (table 7). The percent discrepancies of the flow simulation volumetric water budget for non-pumping conditions and pumping conditions were both 0.01.

Sensitivity Analysis

Model parameters were adjusted, and the simulations were rerun to determine model sensitivity to individual parameters. The results of the simulations indicate that the model is most sensitive to global changes in hydraulic conductivity and recharge (table 8). Decreasing the hydraulic conductivity by a factor of 2 resulted in simulated water-level changes up to 11.2 ft in the vicinity of the Test Stands under non-pumping conditions and up to 11.4 ft under pumping conditions. Multiplying the recharge by 0.667 resulted in simulated water-level changes at target wells up to 4.8 ft under non-pumping conditions and up to 6.1 ft under pumping conditions. These changes also produced the largest shifts in RMSE of the modeling simulations (table 8).

Increases and decreases in drain, river, and general head boundary conductances resulted in simulated water-level changes of less than 1 ft at observation wells. Similarly, increasing or decreasing the hydraulic conductivity of the ground-water depression near well P-02 (fig. 9) produced less than 1 ft of change in simulated water levels. The lack of model sensitivity to these parameters indicates that minor differences between these assumed values and the actual values probably have no effect on the model solution.

The model sensitivity to pumping rates also was tested because some of the measured pumping rates from previous site records appeared to have slight transcription errors. Rerunning the model using the measured pumping rates as the simulated pumping resulted in water-level changes of 0.7 ft or less near or downgradient from the pumped wells and less than 0.1 ft elsewhere (not shown in table 8). The relative lack of model sensitivity to the pumping rates was further shown by doubling the pumping rates and rerunning the model, which resulted in 1 to 4 ft of water-level change near the pumped wells, but less than 0.5 ft of water-level change elsewhere.

Solute-Transport Model

The solute-transport model (MT3D) was used to simulate migration of benzene associated with the diesel-fuel spill at the northeastern part of the Test Stands building. The MODFLOW simulations provided the block-centered finite-difference grid and the ground-water flow field used by the solute-transport simulations. The transport model used a mixed Eulerian-Lagrangian method to solve the advective-dispersive-reactive equation that describes the transport of miscible contaminants in ground water. The modified method of characteristics (MMOC) was used as the solver.

Solute-transport simulation using MT3D requires breaking the simulated time period into time steps, which then are further discretized into transport steps. The non-pumping phase of the solute-transport modeling was simulated using 37 time steps. The pumping phase of the solute-transport model was simulated using 40 time steps. Each time step simulated 30 days of transport. Transport-step discretization was handled internally in the MT3D code.

Table 8.--Sensitivity analysis of the ground-water flow model at a gas-turbine manufacturing facility, Greenville, S.C.

[For the purpose of this table, wells considered in the Test Stands contamination and nearby area are P-01, WQ-02, WQ-03, WQ-04, WQ-05, WQ-06, WQ-09, and WQ-10; Maximum head change values are reported as simulated calibration heads minus simulated sensitivity-analysis heads; RMSE, Root mean squared error; Change in RMSE, Calibrated root mean squared error minus sensitivity-analysis root mean squared error]

		Test Stands contamination and nearby area				**Entire modeled area**				**Change in RMSE (feet)**	
		Non pumping conditions		**Pumping Conditions**		**Non pumping conditions**		**Pumping conditions**			
Parameter	**Change**	**Maximum head change (feet)**	**Well location of maximum head change**	**Maximum head change (feet)**	**Well location of maximum head change**	**Maximum head change (feet)**	**Well location of maximum head change**	**Maximum head change (feet)**	**Well location of maximum head change**	**Non pumping conditions**	**Pumping conditions**
Recharge	x1.33	-4.2	WQ-02	-2.6	WQ-02	-5.3	Several wells	-3.4	WQ-47	-1.65	-1.14
	x.667	4.8	WQ-02	6.1	WQ-02	6.2	WQ-47, WQ-48	7.9	WQ-16, WQ-47	-2.60	-3.05
Drainage conductance	x2	.1	Several wells	.2	WQ-02, WQ-03, WQ-09	.2	Several wells	.3	WQ-38, WQ-43	-.02	.03
	x.5	.1	WQ-02, WQ-03	.1	Several wells	-.2, .2	WQ-19, WQ-38, WQ-48	-.2, .2	Several wells	.00	.01
River conductance	x2	.4	WQ-06	.3	Several wells	.4	P-02, P-09, WQ-06	.4	P-02, P-03, P-09	.00	.06
	x.5	-.5	WQ-06	-.4	WQ-06	-.6	P-02, P-09	-.6	P-02	-.03	-.06
General head boundary conductance	x2	.2	WQ-02	.2	WQ-02, WQ-03	.3	WQ-19, WQ-47, WQ-48	.6	WQ-69	-.03	.02
	x.5	-.1, .1	WQ-02, WQ-06	.1	P-01, WQ-05, WQ-06	-.1, .1	Several wells	-.4	WQ-69	.01	.00

Table 8.--Sensitivity analysis of the ground-water flow model at a gas-turbine manufacturing facility, Greenville, S.C.

[For the purpose of this table, wells considered in the Test Stands contamination and nearby area are P-01, WQ-02, WQ-03, WQ-04, WQ-05, WQ-06, WQ-09, and WQ-10; Maximum head change values are reported as simulated calibration heads minus simulated sensitivity-analysis heads; RMSE, Root mean squared error; Change in RMSE, Calibrated root mean squared error minus sensitivity-analysis root mean squared error]

		Test Stands contamination and nearby area				Entire modeled area				Change in RMSE (feet)	
		Non pumping conditions		Pumping Conditions		Non pumping conditions		Pumping conditions			
Parameter	Change	Maximum head change (feet)	Well location of maximum head change	Maximum head change (feet)	Well location of maximum head change	Maximum head change (feet)	Well location of maximum head change	Maximum head change (feet)	Well location of maximum head change	Non pumping conditions	Pumping conditions
Hydraulic conductivity	x2	7.2	WQ-02	7.5	WQ-02	9.3	WQ-47, WQ-48	9.9	WQ-48	-4.31	-4.01
	x.5	-11.2	WQ-02	-11.4	WQ-02	-13.9	WQ-48	-14.4	WQ-48	-6.38	-7.16
Hydraulic conductivity near well P-02	x2	.3	WQ-06	-.4	WQ-06	.3	P-02, WQ-06, WQ-19	-.5	P-02	-.01	-.02
	x.5	-.4	WQ-06	.3	P-01, WQ-03	-.4	P-02, P-03, WQ-06	.3	Several wells	-.02	.03

Modeled Solute-Transport Properties

The finite-difference grid and model boundaries used by the solute-transport model were identical with those used in the ground-water flow simulations. Simulated potential sinks for contaminants included removal by pumping wells and discharge to river cells, drain cells, and the general-head boundaries.

An additional parameter required by the solute-transport model is aquifer effective porosity. For this investigation, simulated effective porosity was 0.30. This value is in the range of specific yields (0.15 to 0.30) cited for saprolite (Fetter, 1988), and is the effective porosity used by previous investigators of this site (Kubal-Furr and Associates, 1997). The simulated value also was chosen to be slightly higher than the average specific yield (0.27) of 36 saprolite-aquifer samples from an investigation in Dawson County, Georgia (Stewart and others, 1964) because specific yield can underestimate effective porosity due to capillary tension and van der Waals forces.

The simulated distribution of free-phase fuel (fig. 12) was based partly on initial investigations by Law Engineering Testing Company (1984a,b) and adjusted to include downgradient areas to which fuel was transported in subsequent years. These additional areas include the vicinity of well WQ-08, where free-phase fuel was detected in August 1991, and the vicinity of well WQ-11, where free-phase fuel was detected in December 1989 (Sirrine Environmental Consultants, 1991b). To account for potential undetected transport farther downgradient, the simulated area of free-phase fuel distribution was chosen to extend beyond well WQ-11 but not as far as the wells where fuel has not been detected (fig. 12). The aqueous concentration of benzene chosen to represent the free-phase fuel was 344 μg/L, which is the concentration of benzene observed in water after contact with diesel fuel (Dunlap and Beckman, 1988). This value was assigned to 208 model cells in the vicinity of the Test Stands and simulated as a constant concentration (not free to fluctuate).

The simulated distribution of benzene outside the free-phase-fuel area (fig. 12) was chosen to include wells where petroleum hydrocarbons were detected in March 1991 (Robert DeGaetano, Kubal-Furr and Associates, written commun., 1997). This area includes well WQ-09; however, contaminant concentrations in water from well WQ-09 probably are not representative of aquifer concentrations because the top of the well screen is approximately 7.2 ft below the average water-table surface. A uniform value of 48 μg/L was used to simulate the starting dissolved-phase concentration because 43 μg/L of benzene was detected in well WQ-08 in March 1991, and by July 1991, free-phase fuel was present in the well. Moreover, 49.4 μg/L was the highest benzene concentration detected in the Test Stands contamination (February 1997). Thus, a value of approximately 43 μg/L, or slightly higher, is a reasonable value to represent ground-water concentrations near free-phase fuel. The concentration of benzene in these model cells was allowed to change with time. Although lower concentrations than 48 μg/L were measured at several wells, the uniform concentration provided a conservative scenario. Moreover, as will be shown in the discussion on sensitivity analysis, the model showed little sensitivity to starting concentrations located outside of the free-phase fuel area.

Biodegradation of benzene was simulated as part of the attenuation process. The lowest laboratory-derived biodegradation rate for benzene under aerobic conditions was 1.25 per day (table 5). The rate was determined under aerobic conditions because the water in most of the wells contained DO at concentrations greater than 1 mg/L. However, ground water from several wells near the free-phase fuel east of the Test Stands contained methane. Moreover, water from wells P-01, WQ-05, WQ-06, and WQ-09 contained less than 1 mg/L of DO. Thus, it is clear that oxygen has been depleted in some areas and that anaerobic biodegradation of organic compounds is active. The overall biodegradation of the fuel, therefore, is a combination of aerobic and anaerobic activity. Because benzene degradation under anaerobic conditions has not definitively been demonstrated under field conditions, and because only one biodegradation rate can be incorporated into the simulation, a value substantially lower than that characterizing aerobic biodegradation is appropriate for simulating areas near the contaminant source at this site. Model simulations using a range of biodegradation rates lower than 1.25 per day showed that a simulated biodegradation rate of 0.009 per day produced reasonable solutions for benzene transport. Therefore, 0.009 per day was used to simulate benzene biodegradation for the calibrated model.

Adsorption of benzene onto solid particles was not simulated as an attenuation mechanism because laboratory analysis of sediment from the site showed insignificant potential for benzene sorption. Moreover, simulation of benzene movement with no adsorption provides a conservative evaluation of benzene transport.

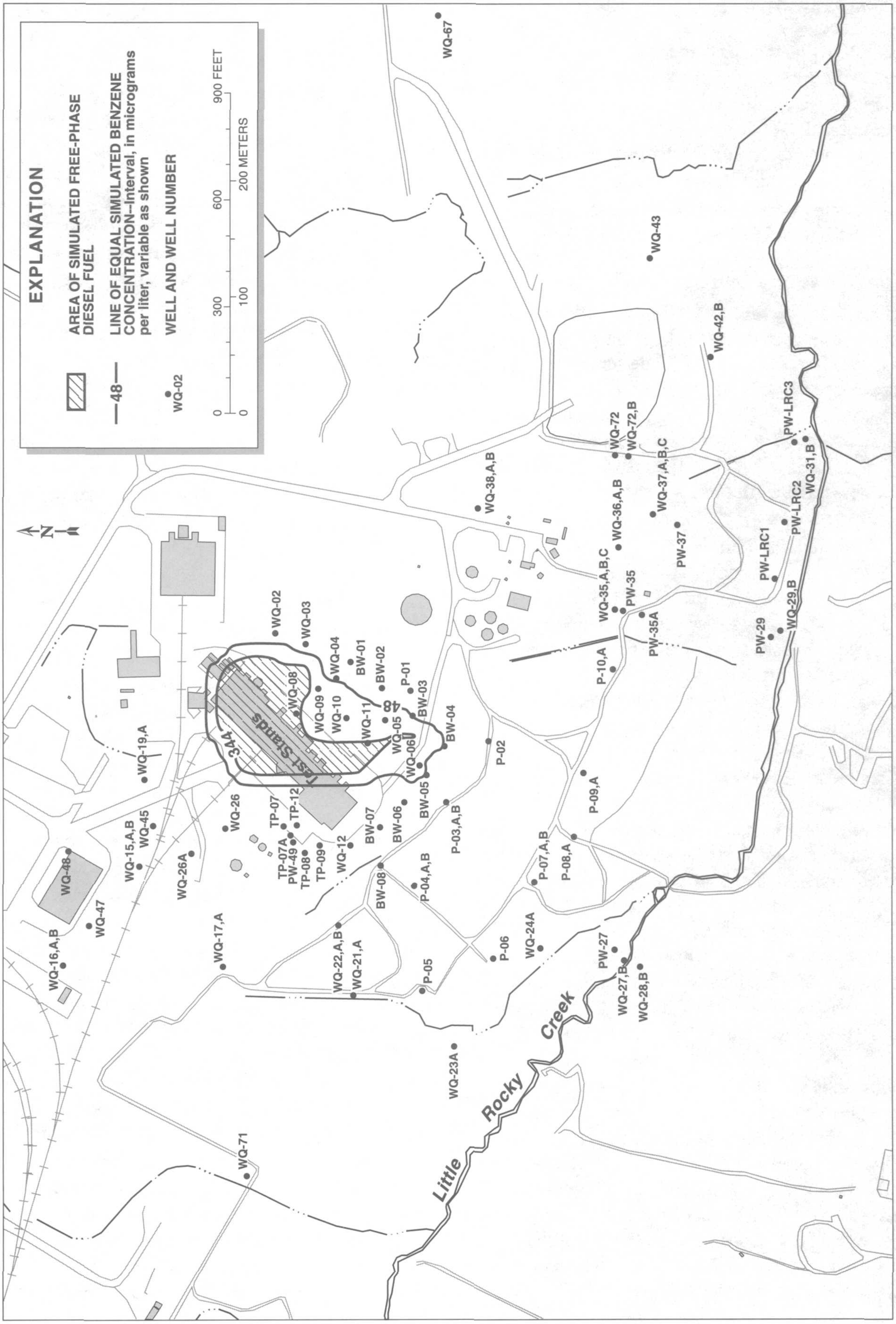


Figure 12. Simulated distribution of free-phase diesel fuel and starting benzene concentrations from the Test Stands contamination at a gas-turbine manufacturing facility, Greenville, S.C.

Diffusion, or the migration of a chemical species as a result of its chemical potential, was incorporated into the solute-transport model by using a uniform starting molecular diffusion coefficient characteristic of other volatile organic compounds, such as trichloroethane and trichloroethene (Lee and Chrysikopoulos, 1995). Thus, 6.278 x 10^{-4} ft^2/d was used to simulate the benzene effective molecular diffusion coefficient (D_m).

Longitudinal dispersivity of benzene, or the mechanical mixing from advective flow in porous media, was initially simulated using a uniform value of 25 ft. During calibration, the value was increased to 50 ft. The ratio of simulated horizontal transverse to longitudinal dispersivity (α_t/α_l) was 0.2, which is within the range of values derived from other literature (Freeze and Cherry, 1979). This combination of longitudinal and transverse dispersivity allowed the simulated contamination to expand laterally as observed in the field.

Calibration Procedure, Criteria, and Results

Calibration of the solute-transport model consisted of simulating transport over two time periods and allowing concentrations of benzene to change with time. The first of the two time periods used hydraulic conditions represented by the ground-water flow nonpumping simulation. This solute-transport simulation represented the period from April 1991 to April 1994. The second time period represented solute transport from June 1994 to July 1997 and used hydraulic conditions from the ground-water-flow pumping simulation.

Simulated August 1997 benzene concentrations were compiled for model cells that approximated the locations of observation wells where benzene concentrations in ground water were measured in August 1997. The RMSE was calculated for 4 wells that contained benzene concentrations greater than the detection limit of 5 µg/L. Simulations producing benzene concentrations that substantially varied from the measured data were considered inaccurate and re-simulated after modifying one of the solute-transport parameters (dispersivity, ratio of horizontal transverse to longitudinal dispersivity, or biodegradation coefficient). The final simulated benzene concentrations from the nonpumping transport simulation were used as starting concentrations for the pumping transport simulation.

The model was considered calibrated when simulated benzene concentrations at wells were approximately the same as measured concentrations at target wells (fig. 13). The simulated benzene concentrations differed by 2 µg/L or less from the measured concentrations at wells with 10-ft well screens. Wells that had 30-ft well screens (wells BW-03 and BW-04) were purposely simulated with higher-than-measured concentrations to account for probable in-well dilution effects from mixing with water across the screened interval during sampling (table 9). Simulated benzene concentrations were below detection limits (5 µg/L) at wells where measured concentrations were below detection limits (wells WQ-06 and P-02). The percent discrepancy of the cumulative mass budget was 1.32 for nonpumping conditions and 1.48 for pumping conditions, which showed good calibration.

Table 9. Comparison of simulated and measured benzene concentrations (August 1997) at a gas-turbine manufacturing facility, Greenville, S.C.

[<, less than; µg/L, micrograms per liter; °,30-ft well screens]

	Observation wells					
	WQ-04	WQ-05	WQ-06	BW-03°	BW-04°	P-02
Measured benzene concentrations (µg/L)	29	16	<5.0	15	8.7	<5.0
Simulated benzene concentrations (µg/L)	30	14	4.3	26	28	3.0

Sensitivity Analysis

The sensitivity of the solute-transport solution to various parameters was tested by individually adjusting the model parameters and comparing the resulting 1997 test-simulation results to the calibrated results (table 10). Aquifer effective porosity was increased by a factor of 1.25 and decreased by a factor of 0.75. The number of time steps in the solute-transport simulation was tested by rerunning the model with double the number of time steps. The effective molecular diffusion coefficient (D_m) was increased and decreased by a factor of 10. The remaining tested parameters were increased by a factor of 2 and decreased by a factor of 0.5. These parameters were biodegradation rate, longitudinal dispersivity, α_t/α_l, the benzene starting concentrations in the source area (zone of free-phase diesel fuel) and in cells outside the source area, hydraulic conductivity, and recharge.

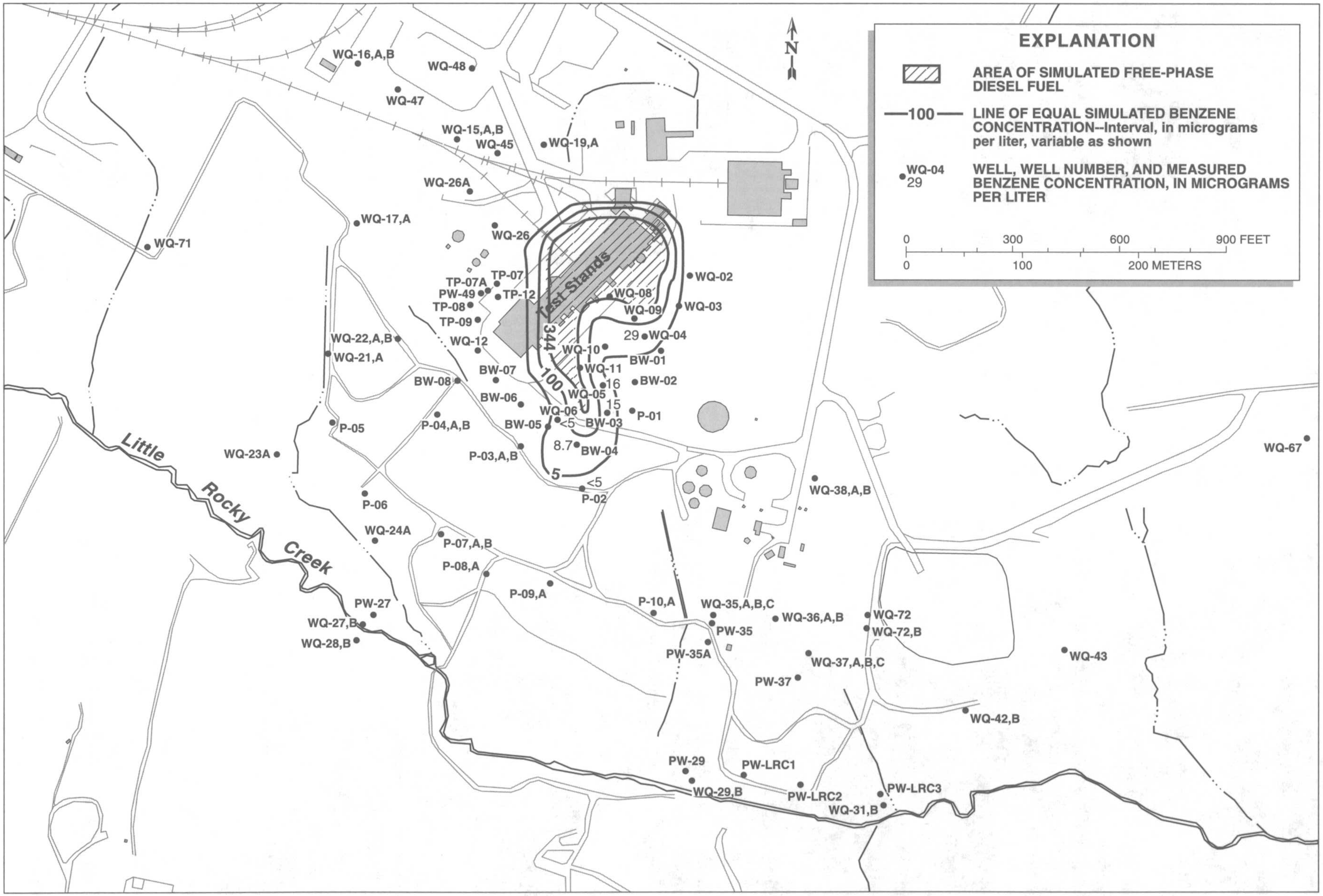


Figure 13. Simulated August 1995 and 2097 benzene distribution (represented by the same contours) and measured August 1997 benzene concentrations from the Test Stands contamination at a gas-turbine manufacturing facility, Greenville, S.C.

Table 10.--Sensitivity analysis of the solute-transport model at a gas-turbine manufacturing facility, Greenville, S.C.

[Change in simulated concentration is defined as the benzene concentration for the calibrated simulation of August 1997 minus the respective concentration derived from the sensitivity simulation; mg/L, milligrams per liter, ft, feet; RMSE, Root mean squared error; α_t/α_l, ratio of horizontal transverse to longitudinal dispersivity; D_m, effective molecular diffusion coefficient]

Parameter	Change	Change in simulated concentration (mg/L)						Minimum distance from leading edge of the contamination to Little Rocky Creek (calibrated distance is 681 ft)	RMSE (mg/L)	Calibrated RMSE-test RMSE (mg/L)
		WQ-04	WQ-05	WQ-06	BW-03	BW-04	P-02			
Biodegradation Rate	x 2	22	6.4	3.3	13	19	2.7	743	11.21	-0.05
	x .5	-46	-12	-9.7	-17	-34	-11	507	38.52	-27.36
Effective porosity	x 1.33	12	3.0	1.9	6.0	10	1.7	718	8.02	3.14
	x .67	-24	-6.0	-4.7	-9.0	-18	-5.2	594	24.66	-13.50
Longitudinal dispersivity	x 2	2.0	-21	-7.7	-28	3.0	.1	656	23.18	-12.02
	x .5	-1.0	9.1	2.8	16	-2.0	-.3	681	12.31	-1.15
α_t/α_l	x 2	2.0	-12	-7.7	-19	1.0	.0	681	18.27	-7.11
	x .5	-1.0	5.6	2.7	10	.0	-.1	668	10.43	.73
Dm	x 2	.0	.0	.0	.0	.0	.0	681	11.16	.00
	x .5	.0	.0	.0	.0	.0	.0	681	11.16	.00
Source, starting concentration	x 2	-30	-14	-4.4	-26	-28	-3.1	619	34.32	-23.16
	x .5	15	6.9	2.1	13	14	1.5	705	8.76	2.40
Cells outside source, starting concentration	x 2	.0	.0	.0	.0	.0	.0	681	11.16	.00
	x .5	.0	.0	.0	.0	.0	.0	681	11.16	.00
Hydraulic conductivity	x 2	-21	-4.0	-2.3	-5.0	-9.0	-2.1	631	19.65	-8.49
	x .5	14	2.0	1.4	5.0	7.0	1.2	705	9.65	1.51
Recharge	x 2	-18	-7.0	-5.7	-11	-22	-6.9	569	25.38	-14.22
	x .5	13	4.5	2.6	10	15	2.4	730	7.17	3.99
Number of time steps	x 2	.0	.0	.0	.0	.0	.0	681	11.16	.00

The solute-transport model appeared to be most sensitive to changes in the biodegradation rate. The simulated August 1997 leading edge of the contamination reached steady-state conditions approximately 62 ft upgradient from the calibrated position when the simulated bioremediation rate was doubled to 0.018 per day. The leading edge reached steady-state conditions approximately 174 ft downgradient from the calibrated position when the simulated biodegradation rate was halved to 0.0045 per day, allowing the simulated contamination to reach approximately 507 ft from Little Rocky Creek by 1997.

An additional influential parameter affecting model sensitivity is the amount of simulated recharge. Under conditions of doubling the calibrated recharge, simulated benzene advectively moved forward to about 569 ft from Little Rocky Creek by 1997 (table 10).

The solute-transport model was also sensitive to effective porosity. Decreasing the effective porosity by about one third allowed the contamination plume to travel an additional 87 ft toward Little Rocky Creek (594 ft from the creek).

The choice of simulated source-area benzene concentrations also affected model sensitivity (table 10). When the simulated concentration of dissolved benzene in the area of free-phase diesel fuel was doubled, the simulated contamination moved forward an additional 62 ft (619 ft from Little Rocky Creek).

The remaining tested parameters showed substantially less influence on model sensitivity. Changes in the dispersivity and in the ratio of horizontal transverse to longitudinal dispersivity affected concentrations at specific wells, but did not influence the distance that the leading edge of the contamination was transported (table 10). The model appeared to be insensitive to changes in the effective molecular diffusion coefficient, starting benzene concentrations in model cells outside of the free-phase fuel area, or doubling the number of time steps in the transport simulation, thus indicating that the calibrated values of these parameters are probably adequate for this investigation.

Predictive Simulations

After the solute-transport model was considered calibrated by approximately matching observed benzene concentrations for August 1997, the calibrated model was run in a predictive mode to simulate 100 years of benzene transport utilizing 200 time steps. The predictive model produced concentrations and areal distribution of contaminants that were the same as those for the August 1997 simulation (fig. 13). The model, therefore, indicates that by August 1997, the benzene contamination had reached approximately steady-state conditions and further transport of benzene was unlikely under the conditions simulated in the model.

The observed lack of change in benzene concentrations at wells WQ-05, WQ-06, and WQ-10 between 1991 and 1996 implies that in some parts of the plume the benzene transport rate is balanced by the benzene removal rate (table 2). The apparent increase in benzene concentration at wells BW-03 and BW-04 between 1991 and 1997 (table 2) may mean that contamination has moved to those wells, or the results may be a sampling artifact related to the distribution of contamination across the 30-ft screened interval.

Model simulations also were run to determine whether benzene would be transported to Little Rocky Creek under a combination of model input parameters constituting a worst-case scenario within the probable limits of model uncertainty. Benzene transport was simulated in a 100-year predictive mode under the combined conditions of reduced biodegradation rate (0.0045 per day), doubled hydraulic conductivity, doubled recharge, and reduced effective porosity (0.2). Using these model input parameters, the simulated contamination reached steady-state conditions within three to five years and produced benzene concentrations at Little Rocky Creek at or near detectable limits of 5 µg/L (fig. 14).

Model Limitations

The model simulations used in this investigation are subject to certain limitations. In general, uncertainties in the input data are propagated to the uncertainties in the model solution. For example, the flow model is primarily sensitive to recharge and hydraulic conductivity. Recharge is measured only with difficulty, and hydraulic conductivity is based on single-well aquifer test measurements as well as best-fit adjustments made during model calibration. Thus, there is uncertainty associated with these values. Moreover, the flow and transport simulations do not represent unique solutions within the realm of available data. Other slightly different configurations and values of recharge and hydraulic conductivity conceivably could produce similar head configurations and acceptable discharge quantities to Little Rocky Creek. Again, this produces uncertainty in the model results. Nevertheless, the values used in this simulation represent the solution producing a reasonable fit of available data. Therefore, the flow simulations probably are valid within the realm of application of this investigation.

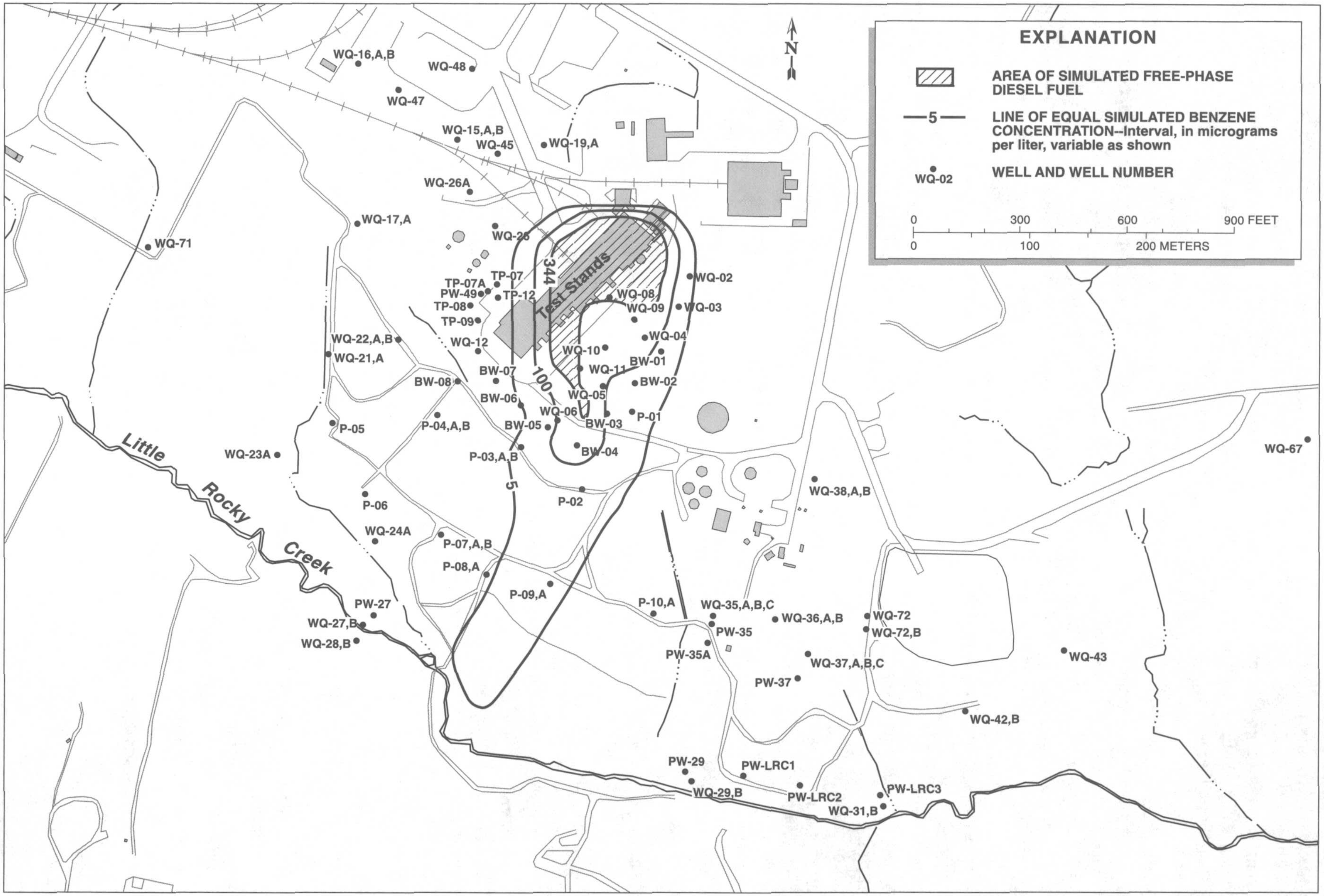


Figure 14. Simulated 2097 benzene distribution from the Test Stands contamination under conditions of reduced biodegradation rate, increased recharge, and increased hydraulic conductivity at a gas-turbine manufacturing facility, Greenville, S.C.

Similarly, the solute-transport model is subject to certain limitations. The transport model was most sensitive to changes in the biodegradation rate. Because the biodegradation rate is not well understood, this imparts an uncertainty to the solution. However, the simulated biodegradation rate was lower than laboratory results, thus simulating a conservative scenario.

There is some uncertainty in the model prediction based on the model assumption that the free-phase fuel constitutes a constant concentration at a fixed location. The 100-year predictive simulation of benzene movement was run using the measured and estimated distribution of free-phase fuel in the 1980's and 1990's. Continued downgradient movement of the free-phase fuel would transport the dissolved part of the contamination forward as well.

Finally, adsorption of benzene was not simulated as a removal mechanism because the limited number of samples examined showed little adsorption potential. If benzene adsorption is a viable mechanism in downgradient areas, the contaminant movement would be slower than predicted.

ASSESSMENT OF NATURAL ATTENUATION

Assessment of the potential for natural attenuation of ground-water contaminants from the Test Stands, the Diesel Pump Station, and the Service Building involved comparing the potential depletion mechanisms to the simulated or measured transport of the target compounds. As will be shown, natural processes probably are adequate to attenuate benzene from the Test Stands contamination, but not the contamination from the Diesel Pump Station or the Service Building.

Test Stands Contamination

Model simulations indicate that ground-water flow in the vicinity of the Test Stands contamination is toward the water-table depression extending from about well WQ-06 toward Little Rocky Creek (fig 10). Solute-transport simulations indicate that the transport velocity of dissolved constituents increases substantially in and near the water-table depression. Thus, if contaminants are allowed to enter the water-table depression, the ability of natural attenuation to mitigate contaminant transport is substantially reduced. In the case of benzene, however, the solute-transport simulation suggests that natural attenuation is sufficient to prevent migration to Little Rocky Creek above concentrations of 5 µg/L. Predictive simulations using calibrated model parameters showed that after 100 years, no detectable benzene was transported to Little Rocky Creek (fig. 13). When the model was rerun using a combination of model input parameters constituting a worst-case scenario, simulated benzene concentrations in ground water at Little Rocky Creek were at or near detectable limits of 5 µg/L (fig. 14). The actual values of the simulated parameters probably allow more benzene attenuation than those modeled by the worst-case scenario, and it is probable that benzene will not be transported to Little Rocky Creek at concentrations greater than 5 µg/L

Decane and naphthalene are two additional compounds of interest at this facility. They are examined qualitatively because the lack of site-specific historical data on these compounds precludes a quantitative approach. Decane is of interest because it is a poorly soluble component of free-phase fuel and was used here to gain an understanding of free-phase fuel migration. Naphthalene is of interest because it is a petroleum hydrocarbon that has been measured in ground water east of the Test Stands. Neither compound is regulated, although there is a recommended South Carolina interim drinking water advisory level for naphthalene of 25 µg/L (Brown, 1996).

The laboratory-derived apparent adsorption coefficient of decane in sediment from the site was 7.4×10^{-8} cubic ft per milligram (ft^3/mg). Although the measured adsorption coefficient for decane includes both adsorption and hydrophobic exsolution, the net effect on contamination migration is the same: a substantial retardation of decane migration relative to benzene. These data support the observation that migration of the free-phase fuel appears to be substantially slower than benzene and other dissolved constituents.

Naphthalene adsorption was not measured in this investigation. However, a generalized conceptualization of naphthalene adsorption can be obtained from published investigations. Naphthalene adsorption in silty sand measured in a previous investigation was 4.84×10^{-6} ft^3/mg (Campbell and others, 1996). By comparison, the adsorption coefficient for toluene in the same investigation was substantially lower (3.32×10^{-8} ft^3/mg). Comparison of naphthalene to toluene adsorption behavior can provide a qualitative comparison of naphthalene to benzene adsorption because toluene and benzene are similarly structured. Thus, substantially more naphthalene adsorption would be expected relative to benzene adsorption.

Reasonable approximations of the probable naphthalene and decane (or free-phase fuel) transport distance cannot be ascertained because they are affected by a variety of factors. Although decane and naphthalene are transported more slowly than benzene (by virtue of their relative adsorptive differences), microbial degradation of these compounds is slow under some conditions (Milelcic and Luthy, 1988; Landmeyer and others, 1998). Thus, there is a potential for moderately soluble compounds, such as naphthalene, to be more persistent than benzene.

Service Building and Diesel Pump Station Contamination

Ground-water chemical data from areas downgradient from the Service Building and from the Diesel Pump Station show that a substantial amount of CVOC dechlorination has taken place. Most of the dechlorination appears to have taken place in zones where the CVOC contamination commingled with petroleum hydrocarbon contamination.

Dechlorination in these zones of maximum degradation potential, however, is insufficient to prevent PCE migration into downgradient areas of limited PCE-degradation potential. For example, despite geochemical evidence for substantial dechlorination in the Service Building contamination at well WQ-45, greater than 1,000 µg/L of PCE was still present in ground water at downgradient well WQ-26 (Robert DeGaetano, Kubal-Furr and Associates, written commun., 1997). Moreover, the presence of 790 µg/L of PCE (October 1995) in aerobic ground water in the deeper part of the flow system at well WQ-15 shows that the PCE has been transported into a zone where limited dechlorination is expected. Similarly, the presence of 23.5 µg/L of PCE (October 1995) in aerobic ground water at well P-04, downgradient from a zone of mixed contamination near well TP-07 and near the Diesel Pump Station, shows that PCE has been transported into a horizon where further dechlorination probably is limited. These data show that the commingling of PCE with oxidizable petroleum hydrocarbons in the aquifer can accelerate biodegradation rates, but not sufficiently to mitigate PCE transport. This consideration, with support of recent analyses (Kubal-Furr and Associates, 1996) showing that chlorinated-solvent contamination has been transported at least to wells adjacent to Little Rocky Creek, indicates that natural attenuation is not a suitable remediation approach for contamination from the Diesel Pump Station or the Service Building.

By analogy, these data can be used to evaluate a potential engineered remediation approach to optimize biodegradation rates in source areas by adding oxidizable organic substrate as an electron acceptor for chlorinated aliphatic compound degradation. The data show that adding oxidizable organic substrate would result in a substantial increase in the degradation rates for higher chlorinated solvents such as PCE. However, the fact that PCE continues to be transported out of the zone where this organic substrate is readily available implies that advective factors are significant enough to transport PCE from the treatment area faster than it can be degraded. The implication is that such an engineered approach to remediation would be inadequate to completely prevent transport of PCE in this aquifer system.

CONCLUSIONS

The USGS, in cooperation with the South Carolina Department of Natural Resources — Water Resources Division, investigated petroleum-hydrocarbon and CVOC contamination in ground-water at a gas-turbine manufacturing facility in Greenville, S.C. The contaminants originated from a variety of source areas, and dissolved contaminants were being transported toward Little Rocky Creek, an area of potential point of contact with human or wildlife populations. The purpose of the investigation was to evaluate the natural-attenuation potential for these contaminants. The evaluation utilized field and laboratory data collected during this investigation in combination with data collected from the site during previous investigations. Examination of benzene migration involved integrating hydrogeologic, geochemical, and microbial parameters in a ground-water flow and solute-transport model.

Results of model simulations indicated that ground-water flow in the vicinity of the Test Stands contamination is toward the water-table depression extending from about well BW-04 toward Little Rocky Creek. Solute-transport simulation results indicated that the transport velocity of dissolved constituents substantially increased in and near the water-table depression. Natural attenuation processes, however, appear to be sufficient to prevent migration of benzene to Little Rocky Creek in concentrations greater than detectable limits of 5 μg/L.

The potential transport of naphthalene, a dissolved component of the petroleum-hydrocarbon contamination, could not be quantitatively determined because site-specific historical data are sparse. Although naphthalene is more readily sorbed than benzene, it has been reported to be poorly biodegradable in some other investigations. Thus, the potential for natural attenuation of naphthalene cannot be determined with certainty without additional data. Similarly, the movement of the free-phase diesel fuel east of the Test Stands could not be quantitatively simulated because of limited available data on the extent of contamination and the rate of movement.

Examination of ground-water chemical data from areas downgradient from the Service Building and from the Diesel Pump Station show that a substantial amount of CVOC dechlorination has taken place. Most of the dechlorination appears to have taken place in zones where the CVOC contamination commingled with petroleum hydrocarbon contamination. Dechlorination in these zones of maximum degradation potential, however, is insufficient to prevent PCE migration into downgradient areas of limited PCE-degradation potential. This consideration, supported by recent analyses showing that chlorinated-solvent contamination has been transported from the Pump Station or Service Building at least to wells adjacent to Little Rocky Creek, indicates that natural attenuation is not a suitable alternative for contamination from the Diesel Pump Station or the Service Building. The data also suggest that an engineered attempt to optimize biodegradation rates in source areas by allowing the CVOC's to mix with oxidizable organic substrate would enhance biodegradation, but would be inadequate to completely prevent transport of PCE in this aquifer system.

REFERENCES

BOSS International, 1996, Users Manual for BOSS GMS, BOSS International and Brigham Young University, Madison, WI, 774 p.

Bradley, P.M., and Chapelle, F.H., 1996, Anaerobic mineralization of vinyl chloride in Fe(III)-reducing, aquifer sediments: Environmental Science and Technology, v. 30, p. 2084-2086.

Brown, J. F., 1996, Proposed SC interim drinking water advisory levels for polycyclic aromatic hydrocarbons: Internal Correspondence from South Carolina Department of Health and Environmental Control to Lori Murtaugh, South Carolina Bureau of Drinking Water Protection, 1 p.

Campbell, B.G., Petkewich, M.D., Landmeyer, J.E., and Chapelle, F.H., 1996, Geology, hydrogeology, and potential of intrinsic bioremediation at the National Park Service Dockside II site and adjacent areas, Charleston, South Carolina, 1993-94: U.S. Geological Survey Water-Resources Investigations Report 96-4170, 69 p.

Chapelle, F.H., and McMahon, P.B., 1991, Geochemistry of dissolved inorganic carbon in a coastal plain aquifer: Sulfate from confining beds as an oxidant in microbial CO_2 production: Journal of Hydrology, v. 127, p. 85-108.

Chapelle, F.H., and Lovley, D.R., 1990, Hydrogen concentrations in ground water as an indicator of bacterial processes in deep aquifer systems: *in* Proceedings of the First International Symposium on the Microbiology of the Deep Subsurface: U.S. Environmental Protection Agency, Orlando, Florida, p. 23.

_____ 1992, Competitive exclusion of sulfate reduction by Fe(III)-reducing bacteria: A mechanism for producing discrete zones of high-iron ground water: Ground Water, v. 30, p. 29-36.

Conrad, Ralf, Lupton, F.S., and Zeikus, J.G., 1987, Hydrogen metabolism in a mildly acidic lake sediment (Lake Mendota): FEMS Microbiology and Ecology, v. 45, p. 107-115.

Daniel, C.C. III, 1996, Ground-water recharge to the regolith-fractured crystalline rock aquifer system, Orange County, North Carolina: U.S. Geological Survey Water-Resources Investigations Report 96-4220, 59 p.

Davis, J.W., and Carpenter, C.L., 1990, Aerobic biodegradation of vinyl chloride in groundwater samples: Applied and Environmental Microbiology, v. 56, p. 3870-3880.

Davis and Floyd, Inc., 1994, Ground-water quality assessment report, Service Building area, General Electric Company Gas Turbine Manufacturing Facility, Greenville, South Carolina: Consultants report to the General Electric Company, 24 p.

Dunlap, L.E., and Beckman, D.D., 1988, Soluble hydrocarbons analysis from kerosene/diesel type hydrocarbons: *in* Proceedings of the Conference on Petroleum Hydrocarbons and Organic Chemicals in Ground Water: Prevention, Detection and Restoration: November 9-11, 1988: Houston, Texas, p. 37-45.

Fetter, C.W., 1988, Applied Hydrogeology (2^{nd} ed.): Columbus, Ohio, Merrill Publishing Company, 592 p.

Freeze, R.A., and Cherry, J.A., 1979, Ground Water: New Jersey, Prentice-Hall, Inc., 604 p.

Ghoshal, Subhasis, Ramaswami, Anuradha, and Luthy, R.G., 1996, Biodegradation of naphthalene from coal tar and heptamethylnonane in mixed batch systems: Environmental Science and Technology, v. 30, p. 1282 - 1291.

Hach Company, 1983, Water Analysis Handbook: Loveland, Colorado, Hach Company, 689 p.

Hartman, S., De Bont, J.A.M., Tramper, J., Luyben, K.C.A.M., 1985, Bacterial degradation of vinyl chloride: Biotechnology Letters, v. 7, p. 383-388.

Kessler, K.J., and Cross, Tom, 1986, Results of aquifer test, General Electric Company: Interoffice memorandum, Law Engineering Testing Company, May 28, 1986, 5 p.

Koch, N.C., 1968, Ground-water resources of Greenville County, South Carolina: South Carolina State Development Board, Bulletin No. 38, 47 p.

Kubal-Furr and Associates, 1996, 1995 Annual ground-water monitoring report, Consultants report to General Electric Company, 35 p.

_____ 1997, Semiannual ground-water monitoring report, first and second quarter 1997, August 1997: Consultants report to General Electric Company, 25 p.

Landmeyer, J.E., Chapelle, F.H., Petkewich, M.D., and Bradley, P.M., 1998, Assessment of natural attenuation of aromatic hydrocarbons in ground water near a former manufactured-gas plant, South Carolina, USA: Environmental Geology, v. 34, no. 4, p. 279-292.

Law Engineering Testing Company, 1984a, Report of an investigation of fuel spill, General Electric Gas Turbine String Plant, Greenville, South Carolina: Consultant's report to General Electric Company, June 18, 1984, 5 p.

_____ 1984b, Ground-water and oil-level monitoring, General Electric Gas Turbine Plant, Greenville, South Carolina: Consultant's report to General Electric Company, December 12, 1984, 4 p.

_____ 1985, Ground-water and oil-level monitoring, General Electric Gas Turbine Plant, Greenville, South Carolina: Consultant's data-table report to General Electric Company, 1 p.

Lee, K.Y., and Chrysikopoulos, C.V., 1995, Numerical modeling of three-dimensional contaminant migration from dissolution of multicomponent NAPL pools in saturated porous media: Environmental Geology, v. 26, p. 157-165.

Lovley, D.R., and Goodwin, Steve, 1988, Hydrogen concentrations as an indicator of the predominant terminal electron-accepting reactions in aquatic sediments: Geochimica et Cosmochimica Acta, v. 52, p. 2993-3003.

McDonald, M.G., and Harbaugh, A.W., 1988, A modular three-dimensional finite-difference ground-water flow model: Techniques of Water-Resources Investigations of the U.S. Geological Survey, book 6, chapter A1, 586 p.

Milelcic, J.R., Luthy, R.G., 1988, Degradation of polycyclic aromatic hydrocarbon compounds under various redox conditions in soil-water systems: Applied Environmental Microbiology, v. 54, p. 1182 - 1187.

National Oceanic and Atmospheric Administration, 1989-96, Climatological data annual summary, South Carolina, volumes 88-99.

Nutter, L.J., and Otton, E.G., 1969, Ground-water occurrence in the Maryland Piedmont: Maryland Geological Survey Report of Investigations no. 10, 56 p.

Phelps, T.J., Malachowsky, K., Schram, R.M., and White, D.C., 1991, Aerobic mineralization of vinyl chloride by a bacterium of the order *Actinomycetales*: Applied and Environmental Microbiology, v. 57, p. 1252-1254.

Rust Environment & Infrastructure, 1994, Wastewater Treatment Plant area interim remedial action extraction system startup: Consultant's report to General Electric Company, 24 p.

_____ 1995a, Fourth quarter and annual 1994 ground-water quality assessment report, General Electric Company, Gas Turbine Manufacturing Facility, Greenville, South Carolina: Consultant's report to General Electric Company, 43 p.

_____ 1995b, Wastewater treatment plant area interim remedial action evaluation report, General Electric Company Gas Turbine Facility, Greenville, South Carolina: Consultants report to the General Electric Company, 11 p.

Schwarzenbach, R.P., and Westall, J., 1981, Transport of nonpolar organic compounds from surface water to ground water - Laboratory sorption studies: Environmental Science and Technology, v. 15, no. 11, p. 11-18.

SEC Donohue, 1992a, Site-wide environmental investigation for the General Electric Company Gas Turbine Facility, Greenville, South Carolina: Consultants report to the General Electric Company, 128 p.

_____ 1992b, Third quarter 1992 Test Stands sampling and analysis report, the General Electric Company Gas Turbine Facility, Greenville, South Carolina: Consultants report to the General Electric Company, 8 p.

Sewell, G.A., and Gibson, S.A., 1991, Stimulation of the reductive dechlorination of tetrachloroethene in anaerobic aquifer microcosms by the addition of toluene: Environmental Science & Technology, v. 25(5), p. 982-984.

Sirrine Environmental Consultants, 1991a, VLF-EM Surface geophysical survey of the General Electric Gas Turbine Manufacturing Facility, Greenville, South Carolina: Consultant's report to General Electric Company, 15 p.

_____ 1991b, Test Stand sampling and analysis, 3rd quarter 1991: Consultant's report to General Electric Company, 9 p.

_____ 1992, Ground-water quality assessment report, General Electric Gas Turbine Manufacturing Facility, Greenville, South Carolina: Consultant's report to General Electric Company, 16 p.

Stewart, J.W., 1962, Water-yielding potential of weathered crystalline rocks at the Georgia Nuclear Laboratory: U.S. Geological Survey Professional Paper 450-B, p. 106-107.

Stewart, J.W., Callahan, J.T., Carter, R.F., and others, 1964, Investigation at the site of the Georgia Nuclear Laboratory, Dawson County, Georgia: U.S. Geological Survey 1133-F, 90 p.

Stookey, L.L., 1970, FerroZine, a new spectrophotometric reagent for iron: Analytical Chemistry, v. 42, p. 779-781.

Stumm, Werner, and Morgan, J.J., 1981, Aquatic chemistry (2d ed.), New York, John Wiley and Sons, 780 p.

U.S. Environmental Protection Agency, 1983, Methods for chemical analysis of water and waste, EPA-600/4-79-020: U.S. Environmental Protection Agency, Washington, D.C.

_____ 1986, Test methods for evaluating solid waste, Physical/chemical methods, SW-846 (3rd ed.): U.S. Environmental Protection Agency, Washington, D.C., 1919 p.

Volkering, F., Breure, A.M., and Van Andel, J.G., 1993, Effect of micro-organisms on the bioavailability and biodegradation of crystalline naphthalene: Applied Microbiology and Biotechnology, v. 40, p. 535-540.

Vroblesky, D.A., and Chapelle, F.H., 1994, Temporal and spatial changes of terminal electron-accepting processes in a petroleum-hydrocarbon-contaminated aquifer and the significance for contaminant biodegradation: Water Resources Research, v. 30, no. 5, p. 1561-1570.

Vroblesky, D.A., Bradley, P.M., and Chapelle, F.H., 1996, Influence of electron donor on the minimum sulfate concentration required for sulfate reduction in a petroleum hydrocarbon-contaminated aquifer: Environmental Science and Technology, v. 30, no. 4, p. 1377-1381.

Vroblesky, D.A., Rhodes, L.C., Robertson, J.F., and Harrigan, J.A., 1996, Locating VOC contamination in a fractured-rock aquifer at the ground-water/surface-water interface using passive vapor collectors: Ground Water, v. 34, no. 2, p. 223-230.

Vroblesky, D.A., Bradley, P.M., Lane, J.W. Jr., and Robertson, J.F., 1997, Transport and transformations of chlorinated-solvent contamination in a saprolite and fractured rock aquifer near a former wastewater-treatment plant, Greenville, South Carolina: U.S. Geological Survey Water-Resources Investigations Report 97-4003, 76 p.

Zheng, C., 1990, MT3D: A modular three-dimensional transport model for simulation of advection, and chemical reactions of contaminants in ground-water systems: S.S. Papadopulus & Associates, Inc., 146 p.