U.S. Department of the Interior
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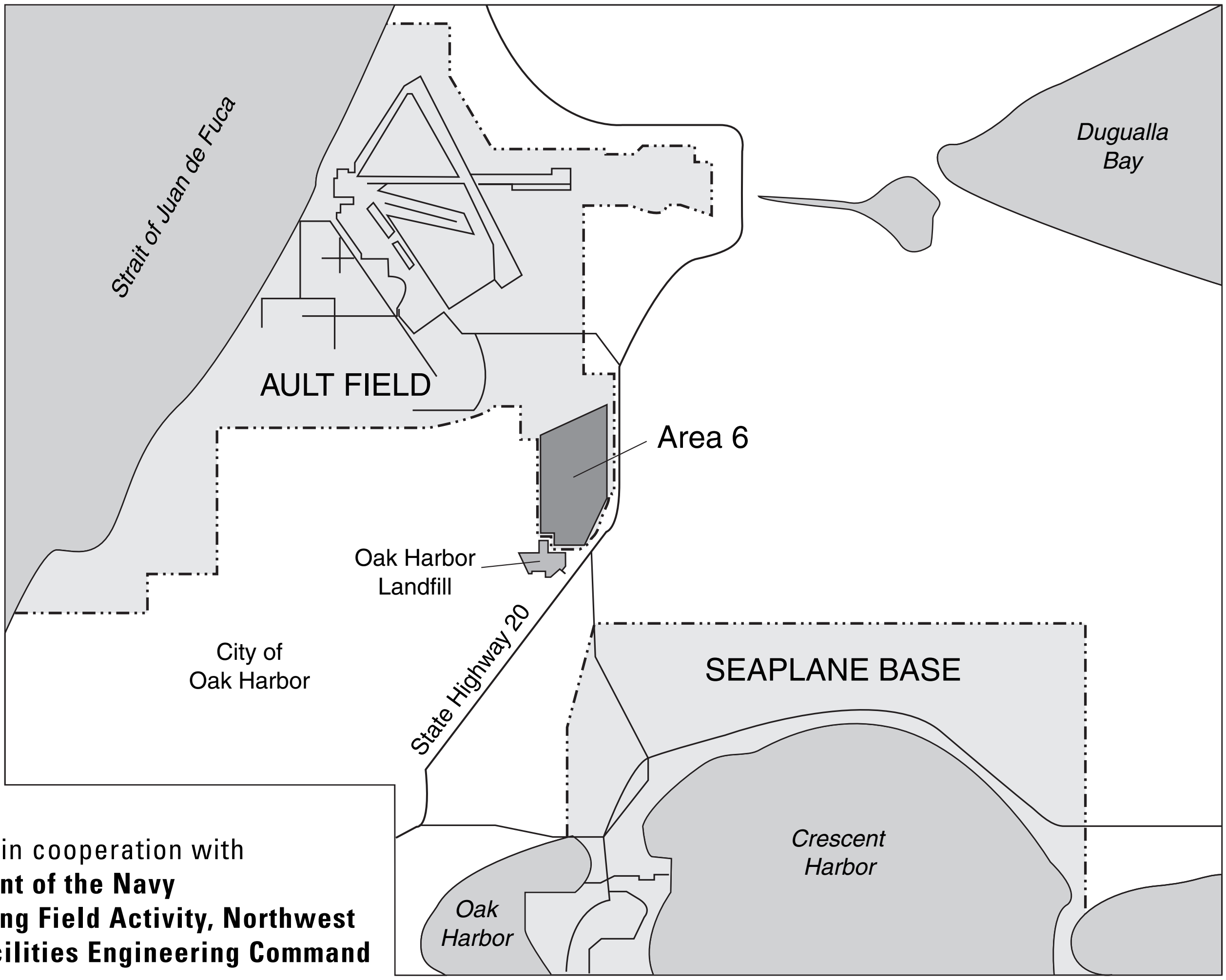
Water-Resources Investigations Report 00-4060



Prepared in cooperation with
Department of the Navy
Engineering Field Activity, Northwest
Naval Facilities Engineering Command

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By R.S. Dinicola, S.E. Cox, *and* P.M. Bradley

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Tacoma, Washington
2000

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EXECUTIVE SUMMARY

PURPOSE, SCOPE AND METHODS

The U.S. Geological Survey evaluated the effectiveness of natural attenuation at meeting remediation objectives for chlorinated volatile organic compounds (VOC's) in shallow ground water at Area 6, Naval Air Station (NAS) Whidbey Island. The VOC's of concern at the site are 1,1,1-trichloroethane (TCA), trichloroethene (TCE), 1,2-dichloroethane (DCA), *cis*-1,2-dichloroethene (*cis*DCE), 1,1-dichloroethene (DCE), and vinyl chloride (VC). The effectiveness of natural attenuation without utilization of the existing pump and treat system was evaluated. The evaluation considered changes in contaminant concentrations over time, ground-water chemistry evidence for contaminant degradation at the site, and results from laboratory experiments demonstrating the potential for degradation of selected compounds in Area 6 aquifer materials under ambient conditions. The Domenico (1987) analytical flow and transport model embedded within the BIOSCREEN program was used to evaluate the effectiveness of natural attenuation for controlling downgradient contaminant migration. Data used in the investigation included contaminant chemistry and hydrogeological data collected by the Navy and the City of Oak Harbor between 1989 and 1998, and ground-water geochemistry data collected by the USGS in 1997 and 1998.

EVIDENCE FOR NATURAL ATTENUATION

In the southern contaminant plume at Area 6 where vinyl chloride (VC) is the primary contaminant of concern, natural attenuation has been effective at controlling the migration of contaminants disposed of in the Navy landfill. Contaminant concentrations have generally remained unchanged at relatively low values both before and after the onset of pump and treat remediation. Contaminant degradation has been and continues to be substantial in the southern plume. Contaminant degradation products are more abundant than the parent compounds TCA and TCE that were originally disposed of at Area 6. Oxidation-reduction (redox) conditions in ground water are very favorable for both reductive dechlorination of tri- and dichlorinated VOC's, and for subsequent mineralization of any remaining dichlorinated compounds and VC. Changes in concentrations of redox reactants and products along a ground-water flow path through the southern plume confirm that conditions are favorable for the inferred degradation processes.

In the western contaminant plume where TCA, TCE, and selected degradation products are the primary contaminants of concern, natural attenuation has not been as effective at controlling the migration of contaminants disposed of near the former hazardous waste storage area. Changes in contaminant concentration over time indicate substantial down-gradient transport of contaminants from the original source area to at least the southern site boundary. Reductive dechlorination of TCA and TCE currently is not substantial in the upper part of the aquifer, but is more substantial in the deeper part of the aquifer. Abiotic degradation of TCA is occurring throughout the aquifer. Parent compounds are more abundant than intermediate degradation products in the shallow part of the aquifer, but the opposite is true in the deeper part of the aquifer. Redox conditions are not favorable for reductive dechlorination of TCA and TCE in the upper part of the aquifer, but they are more favorable at depth. Changes in concentrations of redox reactants and products along a shallow and a deep ground-water flow path through the western plume confirm the inferred degradation processes.

Laboratory microcosm experiments demonstrated that microorganisms indigenous to the shallow aquifer at Area 6 are capable of significant mineralization of VC and DCA under either aerobic or mildly reducing conditions. The VC results and DCA results for aerobic conditions had been demonstrated in previous investigations, but the present study is the first to demonstrate the mineralization of DCA coupled to manganese reduction.

EFFECTIVENESS OF NATURAL ATTENUATION

Natural attenuation alone could likely achieve all remediation objectives for the southern contaminant plume, with the exception that the cessation of pump and treat would allow some off-base migration of small amounts of VC. However, subsequent mineralization of VC would keep the southern plume from expanding beyond the boundaries of the adjacent City of Oak Harbor landfill, so risks to current and future downgradient ground water would not increase if natural attenuation alone were used to limit plume migration. It is not known how long it will take for either natural attenuation or pump and treat to reduce VC concentrations to below cleanup standards throughout the entire southern plume area. Once the landfill cap successfully controls the migration of additional contaminants to the southern plume, mineralization of VC could then reduce plume concentrations to below cleanup levels. The pump and treat system has not had an observable effect on reducing contaminant concentrations or mass in the southern plume.

In contrast to the southern plume, natural attenuation alone could not currently achieve the remediation objectives for the western contaminant plume. The pump and treat system appears to be more effective at limiting plume migration and at removing TCA and TCE from shallow ground water. Neither natural attenuation nor pump and treat have prevented all migration of VOC's across the site boundary, particularly to the west of the Navy property boundary. Natural attenuation has reduced contaminant concentrations and mass in the western plume. Immediate cessation of pump and treat in the western plume would allow the existing plume to spread southward off-base, and cleanup standards for at least TCA and DCE would likely be exceeded in a few years at downgradient locations that could pose a risk to potential ground-water users.

CONCLUSIONS

Natural attenuation is a viable alternative to pump and treat for meeting remediation objectives in the vicinity of the southern contaminant plume. The combination of historically low contaminant concentrations in ground water, a landfill cap that limits source area contributions, favorable conditions for degradation of VC, and a relatively long downgradient distance to potential receptors are all favorable for natural attenuation as a remediation alternative. Natural attenuation could effectively meet all but one remediation goal that extraction wells PW-2, PW-4, PW-6, PW-7, PW-8 and PW-9 are currently being employed to meet. The goal of preventing migration of all VC across the site boundary could not be met by natural attenuation. Some VC would migrate south of the Navy boundary, but the potential for subsequent VC mineralization downgradient of the base and the existing institutional controls would result in minimal additional risk from using natural attenuation.

In the western contaminant plume natural attenuation is not currently a viable alternative to pump and treat for meeting remediation objectives. There is a possibility that rates for reductive dechlorination of TCE and TCA could increase substantially if the plume was allowed to migrate beneath the Oak Harbor landfill, but there are not enough data to be certain of such an increase. TCA and TCE concentrations at the leading edge of the western plume need to be reduced to at least 25 and 40 µg/L, respectively, to be protective of potential downgradient receptors. Source area TCA and TCE concentrations have decreased substantially over the past ten years, and the extraction wells PW-3 and PW-5 in particular are removing a significant mass of contaminants from ground water, so natural attenuation may be a viable alternative for the western plume in the future.

There would be some possible side benefits of using natural attenuation as an alternative to pump and treat in the southern contaminant plume. The first would be that the resulting decrease in the amount of treated water that would need to be recharged in the swale north of the landfill would result in less off-base migration of contamination across the western site boundary. The second benefit would be a substantial reduction in the amount of dissolved iron and manganese being extracted from the shallow aquifer and run through the treatment system. Removing that source of operation and maintenance problems would result in more effective containment and removal of contamination in the western contaminant plume.

The most critical data gap identified in this evaluation is the paucity of contaminant chemistry information downgradient of the Navy boundary in the vicinity of the southern contaminant plume. Without such data, the behavior of the plume and the protectiveness of natural attenuation to downgradient receptors cannot be verified, and field attenuation rates for VC cannot be determined directly. The existing long-term monitoring plan would need to be reviewed and revised if natural attenuation is selected as a remedy for the contamination in the southern plume. In particular, additional performance monitoring wells may be required downgradient of the property.

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CONVERSION FACTORS, VERTICAL DATUM AND ABBREVIATIONS

Multiply	By	To obtain
inch (in)	25.4	millimeter
foot (ft)	0.3048	meter
acre	0.4047	hectare
	4,047	square meter
mile (mi)	1.609	kilometer
gallon (gal)	3.785	liter

Temperature: To convert temperature given in this report in degrees Fahrenheit (°F) to degrees Celsius (°C), use the following equation: °C =5/9(°F-32).

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

Altitude: In this report, "altitude" is measured in feet above sea level.

Abbreviations:

CA	chloroethane
*cis*DCE	*cis*-1,2-dichloroethene
DCA	1,1-dichloroethane
DCE	1,1-dichloroethene
DO	dissolved oxygen
DOH	Washington State Department of Health
dpm	disintegrations per minute
EDTA	ethylenediaminetetraacetic acid
EPA	U.S. Environmental Protection Agency
ICP	inductively coupled plasma
MCL	maximum contaminant level
NAS	Naval Air Station
NWQL	National Water Quality Lab
PVC	polyvinyl chloride
RI	Remedial Investigation
ROD	Record of Decision
SC	specific conductivity
TCA	1,1,1-trichloroethane
TCE	trichloroethene
TDS	total dissolved solids
TOC	total organic carbon
USGS	U.S. Geological Survey
VC	vinyl chloride
VOC	volatile organic compound
g	grams
kg	kilogram
kg/L	kilograms per liter
mg/L	milligrams per liter
mL	milliliter
µL	microliter
nmol	nanomoles
µg/L	micrograms per liter
µmol	micromoles
µS/cm	microsiemens per centimeter

Natural Attenuation of Chlorinated Volatile Organic Compounds in Ground Water at Area 6, Naval Air Station Whidbey Island, Washington

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INTRODUCTION

There is a need to evaluate the effectiveness of natural attenuation for controlling the migration and reducing the mass of chlorinated volatile organic compounds (VOC's) in ground water at Area 6, Naval Air Station (NAS) Whidbey Island in Washington. An existing pump and treat ground-water remediation system has been in operation at the site since 1995. Although the current system appears to be controlling contaminant migration (Foster Wheeler Environmental, 1998a), during 1995–98 it had only recovered about 210 gallons of VOC's out of the estimated 400,000 gallons of mixed liquid wastes disposed of at the site. Such low VOC recovery rates are typical for pump and treat systems, and the ability of those systems to fully remediate contaminated ground water has been called into question by the U.S. Environmental Protection Agency (EPA) and others (Keeley, 1989; U.S. Environmental Protection Agency, 1989; Haley and others, 1991, Freeze and Cherry, 1989; Mackay and Cherry, 1989).

Combining pump and treat with monitored natural attenuation may lead to more rapid and cost-effective attainment of remediation goals. The Record of Decision (ROD) for Area 6 (URS Consultants, Inc., 1993a) specified that if progress towards achievement of remediation goals was not apparent, the pump and treat system could be modified, including adoption of new technologies that have been proven to be more effective. Monitored natural attenuation may be such a technology.

Monitored natural attenuation refers to the reliance on natural processes to achieve site specific remedial objectives within a reasonable timeframe (U.S. Environmental Protection Agency, 1997). A variety of natural physical, chemical, and biological processes can act without human intervention to reduce the mass, toxicity, mobility, volume, or concentration of contaminants in ground water. Natural attenuation processes typically occur at all sites to varying degrees depending on the contaminants present and the physical, chemical, and biological characteristics of soil and ground water. Processes most relevant to attenuation of chlorinated VOC's in ground water include destructive processes such as abiotic and biotic degradation, and nondestructive processes such as dispersion, sorption, and volatilization. When considering monitored natural attenuation as a remediation strategy, destructive processes are preferred over nondestructive processes. Because historical data from Area 6 at NAS Whidbey Island show an abundance of chlorinated VOC degradation products, monitored natural attenuation has potential as a remedial alternative at the site.

Purpose and Scope

The purpose of this report is to describe an evaluation of the effectiveness of natural attenuation processes at meeting remediation objectives for chlorinated VOC's in shallow ground water at Area 6, NAS Whidbey Island. The evaluation focused on the effectiveness of natural attenuation independent of the existing pump and treat system. Three primary lines of evidence for evaluating the effectiveness of natural attenuation at the site are presented in this report:

1. Historical ground-water contaminant data showing the degree of plume stabilization and (or) decreases in contaminant concentrations over time,
2. Ground-water chemistry data showing conditions are suitable for particular types of contaminant degradation reactions, and to what degree active degradation is taking place, and
3. Laboratory microcosm studies demonstrating that selected degradation processes (aerobic and anaerobic mineralization of vinyl chloride and dichloroethane) can be mediated by indigenous microorganisms in the ambient ground water at Area 6.

The overall effectiveness of natural attenuation was evaluated by comparing contaminant transport rates to attenuation rates. The comparison was quantified by employing a ground-water flow and transport model that considers the combined effects of advective transport, adsorption, dispersion, and contaminant degradation. Data collection and interpretation for this investigation were focused on the uppermost aquifer beneath Area 6. Some data were collected from deeper aquifers, but natural attenuation of the trace contamination in those deeper aquifers was not evaluated.

Description of the Study Area

Following is a summary description of the study area and its setting. More detailed information can be found in the Remedial Investigation (RI) report for Operable Unit 1, NAS Whidbey Island (URS Consultants, Inc., 1993b). Most of the information presented below is from that report.

Geography and Climate

NAS Whidbey Island is located on Whidbey Island in Island County, Washington, on the eastern end of the Strait of Juan de Fuca (fig. 1). The island is 1 to 10 miles (mi) wide and almost 40 mi long. The topography of Whidbey Island is characterized by rolling uplands 100–300 feet (ft) above sea level with steep bluffs along the coast. NAS Whidbey Island is located on the northern part of the island, just north of the City of Oak Harbor (population about 15,000). Forests cover the largest percentage of the island, while urban and agricultural areas cover the remainder. Land use in the vicinity of Area 6, NAS Whidbey Island is primarily residential, commercial, and open forest or cleared tracts. The former City of Oak Harbor landfill, hereafter referred to as the “Oak Harbor landfill,” is immediately south of Area 6.

Whidbey Island has a temperate marine climate characterized by warm, dry summers and cool, wet winters. Mean annual temperature is about 50 degrees Fahrenheit (°F); January is the coolest month, and August is the warmest. Mean annual precipitation ranges from 18 inches in the northern part of the island to 35 inches in the southern part. Mean annual precipitation for NAS Whidbey Island is 19 inches. Snowfall averages less than 8 inches per year and rainfall is generally not intense. There are no perennial streams draining the study area and most surface-water runoff drains to the north into wetlands near the runways at Ault Field.

Hydrogeologic Setting

Whidbey Island lies within the Puget Sound Lowland, a topographic and structural depression between the Cascade Range on the east, and the Olympic Mountains on the west. Whidbey Island is composed of unconsolidated Pleistocene glacial and interglacial deposits overlying Tertiary and older bedrock (Easterbrook and Anderson, 1968). The unconsolidated deposits range from a few hundred to 3,000 ft thick (Sapik and others, 1988). Surficial deposits consist of glacial till, sand, and gravel. Older glacial and interglacial deposits occur at depth.

Ground water beneath NAS Whidbey Island occurs within a series of aquifers that are composed of permeable sand and gravel layers deposited by glacial melt water, separated by finer-grained glacial silt and clay or interglacial fluvial deposits (URS Consultants, Inc., 1993b). The uppermost 200 ft of deposits beneath Area 6 contain three principal water-bearing units, referred to as the shallow aquifer, the intermediate aquifer, and the deep (or sea-level) aquifer (fig. 2). Localized perched aquifers occur above the glacial till in a few places beneath Area 6. The subsurface materials have been characterized into six hydrogeologic units (URS Consultants, Inc., 1993b; Sapik and others, 1988). Those units are the Vashon till, Vashon advance outwash, and four sub-units of the Whidbey formation.

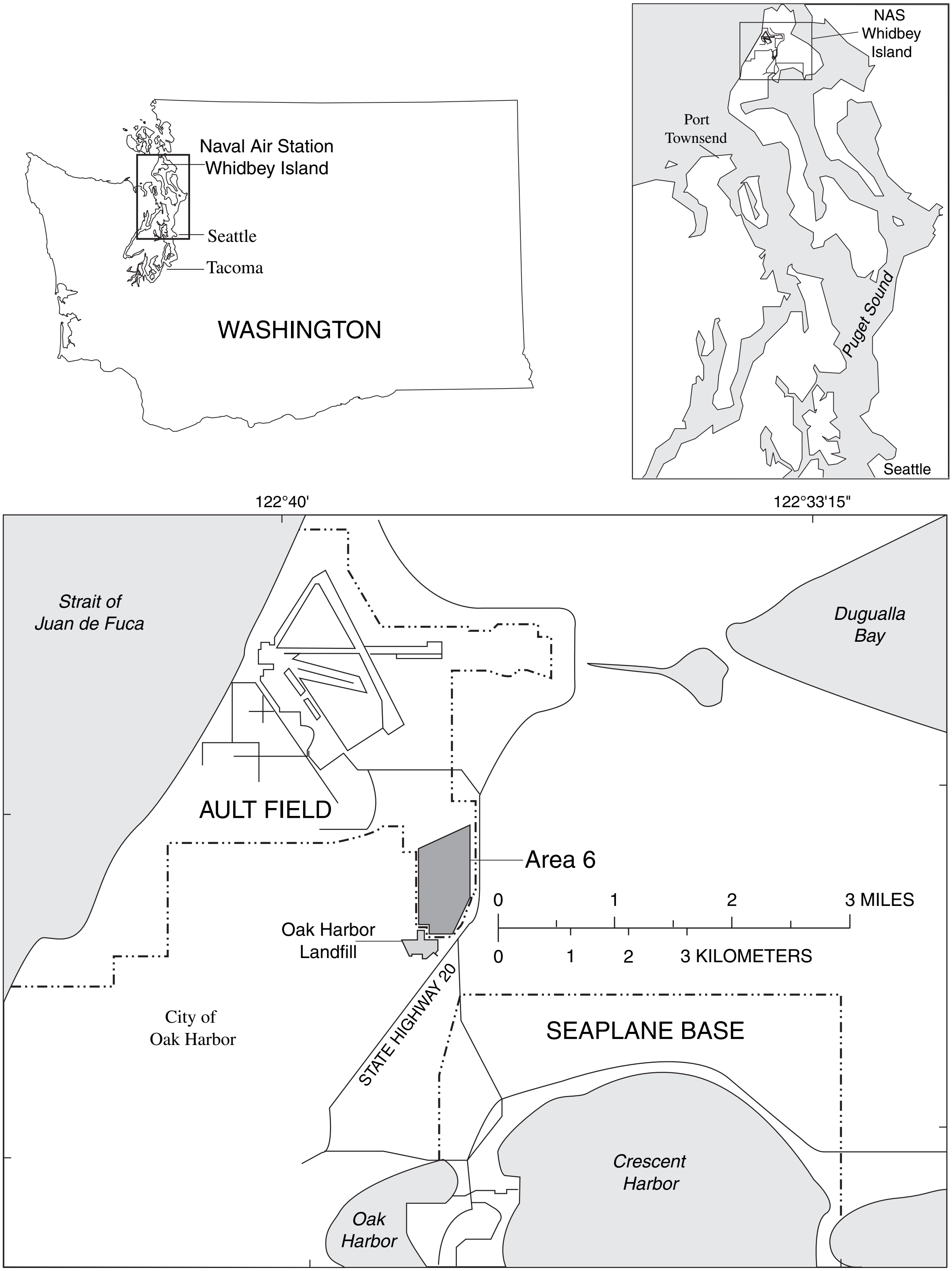

Figure 1. Location of the Area 6 study area, Naval Air Station Whidbey Island, Washington. (From URS Consultants, 1993a.)

The shallow aquifer is contained within the Vashon advance outwash sediments. It is a major water-bearing zone in the region and extends beneath all of Area 6. It has the highest contaminant concentrations beneath Area 6 and was the primary focus of this investigation. The shallow aquifer is generally unconfined, with water levels ranging from about 20 to 145 ft below the ground surface. Ground-water flow in the aquifer is from north to south, and the estimated ground-water flow velocity is about 320 to 640 feet per year (ft/yr). More detailed hydrogeologic characteristics of this aquifer are described in subsequent sections of this report. Eight to 36 ft of fine sands, silts, and clay, with minor peat and wood material, immediately underlie the shallow aquifer. The vertical hydraulic gradient across this underlying confining unit is positive downward at about 0.2 to 0.4 foot per foot (ft/ft).

The intermediate aquifer is contained within a coarse-grained unit of the Whidbey formation. Although not a major water-bearing unit in the region, it extends throughout northern Whidbey Island. Some contaminants have been detected in the intermediate aquifer beneath Area 6, but the concentrations have generally been below levels of concern. The intermediate aquifer is confined, and water levels are generally 5 to 20 ft less than those in the shallow aquifer. Ground-water flow in the intermediate aquifer is from the northwest to the southeast, and the estimated ground-water flow velocity is about 8 to 27 ft/yr. Sixteen to 38 ft of silt and clay, with some sand and woody material immediately underlie the aquifer. The vertical hydraulic gradient across this underlying confining unit is positive downward at about 2.0 to 3.4 ft/ft.

The deep aquifer is contained within another coarse-grained unit of the Whidbey formation. It is a 70- to 75-ft-thick, continuous water-bearing unit in the region. No contaminants have been detected in this aquifer beneath Area 6, with the exception of those resulting from a poorly constructed monitoring well that has been abandoned. The deep aquifer is confined, and water levels range from about 11 to 17 ft above sea level. Ground-water flow in the deep aquifer is generally towards the southwest, and estimated ground-water flow velocity is about 8 to 27 ft/yr. The deep aquifer is underlain by a sequence of at least three more fine-grained units separated by two major water-bearing units, lying atop bedrock.

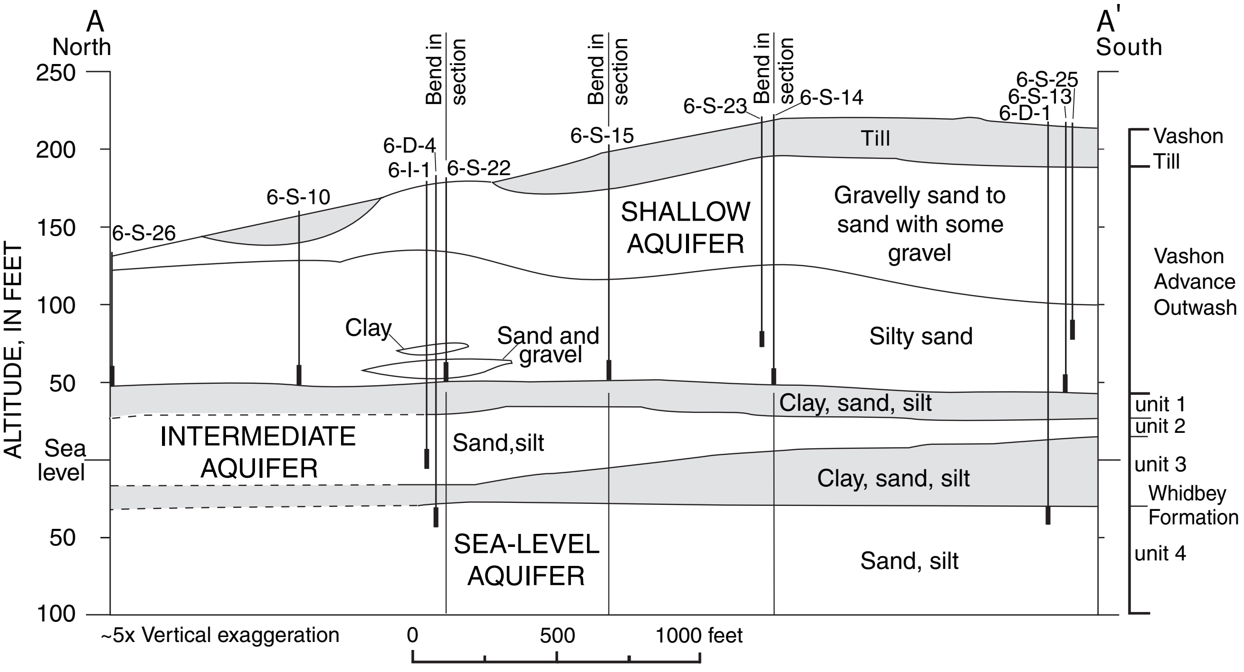

Figure 2. Generalized hydrogeologic section showing stratigraphic units at Area 6. (From URS Consultants, 1993a.)

Contamination and Remediation

NAS Whidbey Island's Ault Field was listed as a Superfund site on the EPA's National Priorities List in February 1990. Area 6 is a 260-acre tract in the southeast corner of Ault Field (fig. 1). Within Area 6 (fig. 3), there are two general locations where wastes were known to have been disposed. A 4.5-acre tract located to the west of the Area 6 landfill included a hazardous waste storage area; an acid pit for disposal of acids, caustics, and solvents; an oily sludge pit; waste oil tanks; and a solvent/caustics waste tank. An unknown quantity of waste was disposed of in the hazardous waste storage area from the early 1980's through 1986. An estimated 300,000 to 700,000 gal of liquid acids, caustics, and solvents were disposed of in the acid pit from the early 1970's to the early 1980's. An estimated 100,000 to 600,000 gal of liquid and sludge was disposed of in the oily sludge pit from 1969 to the mid-1970's.

The 40-acre landfill at Area 6 was primarily used to dispose of Navy household municipal waste from 1969–92. The landfill accepted solid waste, asbestos, wood, rubble, animal remains, construction debris, and hazardous liquid or sludges. The base of the landfill was not lined. Disposal at the landfill began at the northern end and progressed southward. An estimated maximum of 2.2 million gallons of liquids and sludges containing hazardous wastes were reportedly disposed of in the northern two-thirds of the landfill between 1969–83.

The City of Oak Harbor operated a 70-acre landfill from 1953–82 in a former borrow pit located immediately south of Area 6. Approximately 129 tons of dry cleaning wastes were reportedly disposed of in the Oak Harbor landfill, along with domestic wastes, demolition materials, and sewage sludge (Ecology and Environment, 1988). A sewage sludge disposal area was located immediately south of Navy wells PW-4, 6-S-29, and 6-S-19, and a mixed municipal waste disposal area was located about 300 ft south of Navy wells PW-5, PW-6, and PW-7. The site has not been fully characterized, although ground water beneath the Oak Harbor landfill had been monitored regularly at four locations during the 1990's (R. Knudson, City of Oak Harbor, written commun., 1998). Data from that monitoring program are presented in this report when applicable to this investigation.

Ground water beneath Area 6 has been sampled multiple times since 1988. The shallow aquifer is currently contaminated with trichloroethene (TCE), 1,1,1-trichloroethane (TCA), and other VOC's that are probable degradation products of TCE and TCA. Two somewhat distinct contaminant plumes can be identified (fig. 3)—one that begins beneath the former hazardous waste storage area and spreads southward along the western site boundary, and a second that begins beneath the Area 6 landfill and also spreads southward. The western plume is primarily TCE, TCA and their degradation products *cis*-1,2-dichloroethene (*cis*DCE), 1,1-dichloroethane (DCA), and 1,1-dichloroethene (DCE). The southern plume is primarily TCA and the degradation products DCA and vinyl chloride (VC). It was suggested that the VC in the southern plume may have resulted from decomposition of Area 6 landfill debris, presumably polyvinyl chloride (PVC) or other plastics (URS Consultants, Inc., 1993b). However, no evidence from the site or from the literature has been found to support PVC degradation into VC at rates that would result in a detectable ground-water plume. The more likely source for VC in the southern plume is degradation of TCE and TCA, processes well documented in the literature.

Public and private water-supply wells located downgradient of Area 6 have been sampled, and no VOC's connected to NAS Whidbey Island activities were found. In 1989, the Washington Department of Health (DOH) sampled 13 public wells located within a 1-mi radius of Area 6 and the Oak Harbor landfill, and no VOC's were detected. In 1991, DOH again sampled one public and six private wells in the vicinity of Area 6. No evidence of contamination from Area 6 was detected in those wells. As a precautionary measure, the Navy began a program offering voluntary water hookups to the public water supply system for landowners who potentially could be affected, and many domestic wells were subsequently abandoned.

The chosen remedial action for Area 6 (URS Consultants, Inc., 1993a) included capping the Area 6 landfill, and extraction and treatment of contaminated ground water from the shallow aquifer. The extraction wells are labeled with a PW prefix on figure 3. The treated ground water was to be recharged back into the shallow aquifer up-gradient of the extraction wells.

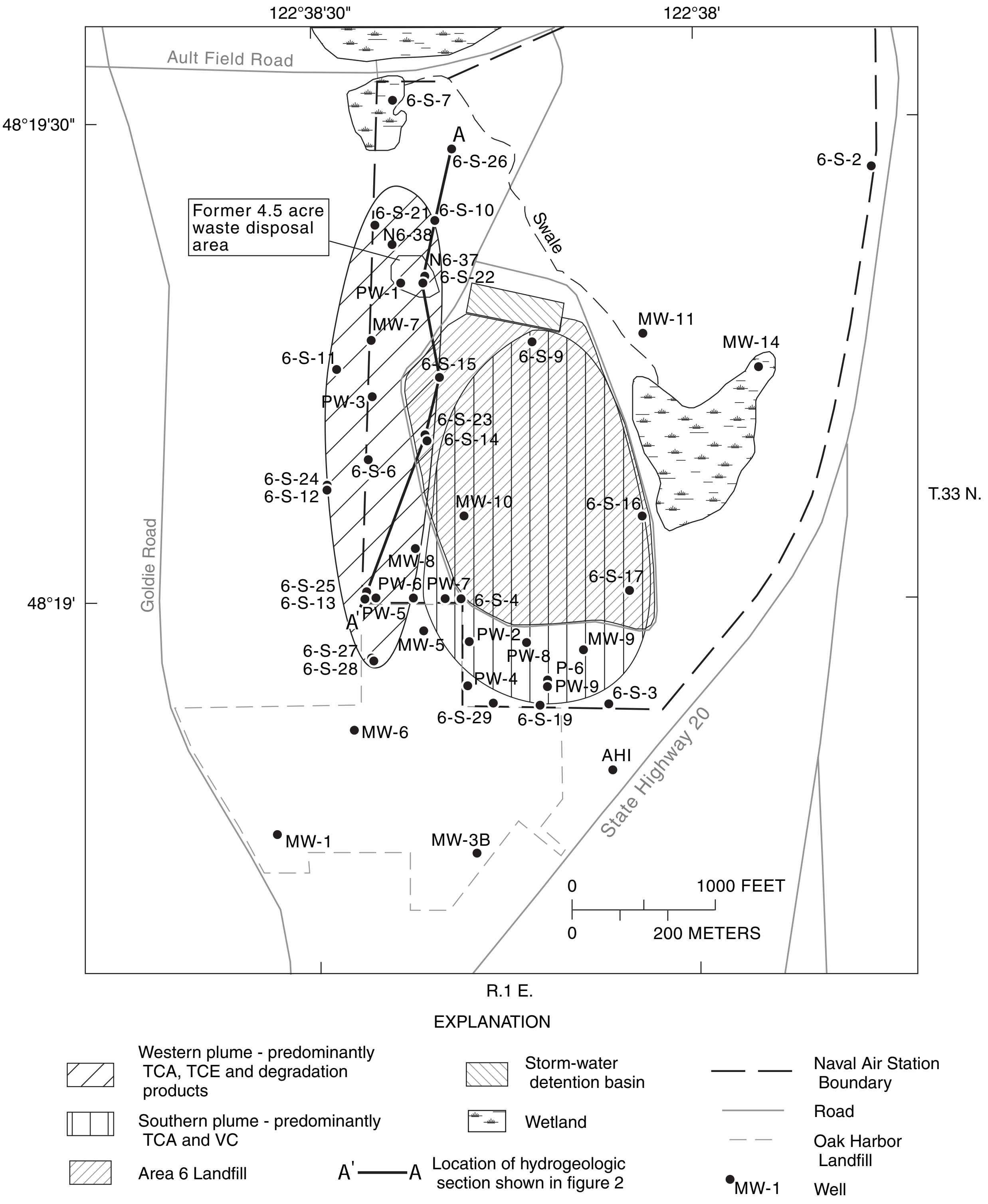

Figure 3. Location of wells, source areas, contaminant plumes, and features in Area 6.

The goals of the remedial action include preventing Area 6 landfill leachate from reaching ground water, preventing further spread of VOC's in ground water, and reducing concentrations of contaminants in the shallow aquifer with the ultimate goal of meeting state and federal drinking water standards at specified points of compliance. The Area 6 landfill cap was completed in 1996, and the pump and treat system began operating July 1995. The pump and treat system has had numerous operational difficulties related to discharging treated ground water through subsurface drains and re-injection wells, and to biofouling of various extraction and treatment equipment (Sonia Murphy, U.S. Navy, Poulsbo, Wash., oral commun, 1998; Hart Crowser, 1999a). Difficulties with recharge of treated water have been mostly overcome by discharging the water into a swale located just northeast of the Area 6 landfill. From there, the water infiltrates and percolates to ground water, mostly in the vicinity of monitoring well 6-S-26. Biofouling problems in extraction wells and treatment equipment still remain, although additional maintenance procedures are being tested (Hart Crowser, 1999b).

Acknowledgments

The authors thank the many people and agencies for their important contributions to this study. Dan Hayes, Sonia Murphy, Douglas Thelin, and John Gordon of the Naval Facilities Engineering Command, Engineering Field Activity Northwest provided guidance and funding for the investigation. Rick Wice of IT Corporation, Tom Goodlin of Foster Wheeler Environmental, and Kathy Lester of EarthWorks Environmental assisted with logistics and field data collection at NAS Whidbey. Jim Landmeyer of the U.S. Geological Survey (USGS) office in Columbia, S.C. assisted with field data collection and microcosm experiments. Russ Knudson of the City of Oak Harbor provided monitoring well data for the Oak Harbor landfill.

BACKGROUND ON REDOX (OXIDATION-REDUCTION) CONDITIONS IN GROUND WATER

Ground-water redox conditions greatly control the occurrence and rate of most contaminant biodegradation reactions. That is because most contaminant degradation processes involve oxidation-reduction (redox) reactions in which electrons are transferred from one compound (the electron donor) to another (the electron acceptor). Organic-carbon compounds, such as those found in natural organic matter, fuel hydrocarbons, certain chlorinated VOC's, and landfill leachate, are the common electron donors that are oxidized in ground water. Various inorganic compounds, such as oxygen, iron (III), and sulfate are the common electron acceptors that are reduced in ground water. Changes in concentrations of electron-donors and acceptors along a ground-water flow path indicate the type of degradation reactions that are occurring. A summary of how ground-water redox conditions are characterized is given here. More information on the topic can be found in works by Wiedemeier and Chappelle (1998), Weidemeier and others (1998), and Weidemeier and others (1996).

Redox conditions in ground water can be defined according to which inorganic compound is acting as the predominant electron acceptor in a given portion of an aquifer, and specific redox conditions are named according to that predominant electron-acceptor. Common redox conditions in ground water are oxygen-reducing (or aerobic), nitrate-reducing, manganese-reducing, iron-reducing, sulfate-reducing, and carbon dioxide-reducing (or methanogenic).

Indigenous microorganisms that require carbon and energy to sustain their growth facilitate the oxidation of organic-carbon compounds in ground water. Different microorganisms are able to use different electron acceptors, and the organisms compete for the available organic-carbon compounds. Oxygen-reducing microorganisms (those that use oxygen as an electron acceptor) can out compete all others when dissolved oxygen (DO) is present. However, if the supply of DO in ground water is depleted and some organic carbon remains, oxygen-reducers will become dormant and nitrate-reducers will predominate until the supply of nitrate is depleted. This sequence continues through manganese-reduction, iron-reduction, sulfate-reduction, and carbon dioxide-reduction. The result of this competitive exclusion is the formation of discrete redox zones in an aquifer that can be identified based on the presence or absence of the different electron acceptors and redox reaction by-products.

The identification of redox conditions can sometimes be deduced from standard chemical analyses of ground water by examining concentrations of various oxidized and reduced inorganic compounds

(Wiedemeier and Chappelle, 1998). The identification of oxygen-reducing (aerobic) conditions is straightforward; if DO is present in ground water at concentrations greater than about 1 milligram per liter (mg/L), then ground water is aerobic. The identification of specific redox conditions in anaerobic ground water (DO concentrations less than 1 mg/L) is more difficult. If nitrate concentrations exceed about 0.5 mg/L, then nitrate reduction is likely. If ground water lacks nitrate, and reduced manganese (Mn (II)) concentrations increase along a ground-water flow path, then manganese reduction is likely. If ground water lacks nitrate, and reduced iron (Fe (II)) concentrations increase along a ground-water flow path, then iron reduction is likely. High concentrations of both Fe (II) and Mn (II) are often found in anaerobic ground water, so it is difficult to distinguish between manganese and iron reduction on the basis of standard chemistry alone. If ground water lacks nitrate, if sulfate (oxidized sulfur) concentrations decrease along a ground-water flow path, and if concentrations of hydrogen sulfide (reduced sulfur) exceed about 0.05 mg/L, then sulfate reduction is likely. Finally, if ground water lacks nitrate, sulfate, and hydrogen sulfide, and methane concentrations are greater than 0.2 mg/L, then carbon dioxide reduction (methanogenesis) is likely.

In practice, there is much uncertainty inherent in the identification of anaerobic redox conditions solely from standard chemistry data. Some of the important redox reaction by-products (Mn(II), Fe(II), and methane in particular) are relatively stable in all anaerobic ground waters, so they can be transported downgradient to areas where they were not originally produced. Also, some other redox reaction by-products (hydrogen sulfide and Fe(II) in particular) readily react to form low-solubility precipitates that are not measurable in ground water.

An alternative method to using standard chemistry is to identify redox conditions through direct measurement of dissolved hydrogen (H_2) concentrations in ground water. Hydrogen is continuously produced and consumed by different microorganisms during anaerobic oxidation of organic matter. It has been demonstrated (Lovely and Goodwin, 1988) that the different microorganisms that facilitate manganese-, iron-, sulfate-, and carbon dioxide-reduction reactions all exhibit different efficiencies in utilizing H_2. Nitrate-reducers are very efficient at utilizing H_2, so they keep H_2 concentrations in ground water at very low levels of less than 0.1 nanomole per liter (nmol/L). Manganese- and iron-reducers use H_2 less efficiently, so they keep H_2 concentrations between 0.1-0.2 and 0.2-0.8 nmol/L, respectively. Sulfate-reducers are less efficient still, so they keep H_2 concentrations at between 1 and 4 nmol/L, and carbon dioxide-reducers are relatively inefficient, resulting in H_2 concentrations greater than 5 nmol/L. The result of this competition for H_2 is that each anaerobic redox condition is indicated by a distinct H_2 concentration in ground water. Thus, H_2 concentration data alone can often be used to identify redox conditions in different locations within an anaerobic aquifer. (H_2 concentrations are not so strictly controlled under aerobic conditions, so DO concentrations must be used to initially distinguish between aerobic and anaerobic conditions.) Although there is some uncertainty in identifying redox conditions from H_2 concentrations alone (primarily due to temperature effects and partial-equilibrium conditions as described by Jakobsen and others, 1998), a combination of standard chemistry and H_2 data can be used to identify redox conditions throughout an aquifer with reasonable confidence.

BACKGROUND ON NATURAL ATTENUATION PROCESSES IN GROUND WATER

Natural attenuation refers to a variety of physical, chemical, and biological processes that act without human intervention to reduce the mass, toxicity, mobility, volume, or concentration of contaminants in soil or ground water (U.S. Environmental Protection Agency, 1997). Processes relevant to natural attenuation of chlorinated VOC's in ground water include destructive processes such as abiotic and biotic degradation, and nondestructive processes such as dispersion, sorption, and volatilization. Destructive processes are preferred when relying on natural attenuation for site remediation (U.S. Environmental Protection Agency, 1997), so contaminant degradation was the primary focus of this investigation. A summary of processes that could naturally attenuate chlorinated ethenes and ethanes in ground water is given here. More information on these processes can be found in works by Wiedemeier and Chapelle (1998), Weidemeier and others (1998), and Weidemeier and others (1996).

Destructive Attenuation Processes

Contaminant degradation reactions are the destructive attenuation processes relevant to VOC's in ground water. Contaminant degradation is the transformation of a chemical compound (the parent compound) into one or more other compounds (the daughter products). Biodegradation reactions are directly mediated by subsurface microorganisms, whereas abiotic degradation reactions are not. Daughter products resulting from degradation of chlorinated VOC's may be more or less toxic than the parent compound, but most intermediate daughter products can be further degraded into nontoxic compounds. From a remediation standpoint, degradation is most desirable when a contaminant is ultimately transformed into innocuous inorganic by-products such as carbon dioxide, water, and chloride.

Over the past two decades, numerous field and laboratory studies have shown that microbes indigenous to ground-water systems can degrade chlorinated ethenes and ethanes. It has also been shown that local subsurface redox conditions are the primary factor controlling the occurrence and rate of biodegradation reactions, although abiotic degradation reactions are often less dependent on redox conditions. The most important degradation reactions for the chlorinated ethenes TCE, DCE, *cis*DCE, and VC, and the chlorinated ethanes TCA, DCA, and chloroethane (CA) are presented here.

Trichloroethene and Other Chlorinated Ethenes

Abiotic degradation of chlorinated ethenes occurs very slowly under conditions commonly found in aquifers (Vogel, 1994). Although considerable discrepancies exist, half-lives for abiotic degradation of chlorinated ethenes have been reported to be as long as 10^{6}–10^{8} years (Jeffers and others, 1989). Abiotic degradation half-lives of TCE can be as short as minutes to a few days for systems amended with abundant zero-valent iron (Gillham and O'Hannesin, 1994), but abundant zero-valent iron is uncommon in natural settings.

Biodegradation of chlorinated ethenes can be very rapid under conditions commonly found in aquifers, and at least two pathways for complete biodegradation of chlorinated ethenes have been demonstrated in field and laboratory studies (fig. 4). Both pathways involve microbially mediated redox reactions. The first pathway consists solely of sequential dechlorination reactions in which the chlorinated ethenes serve as electron acceptors. Trichloroethene can be completely degraded to ethene by this pathway, but the dechlorination usually stops at *cis*DCE or VC in the majority of ground-water systems. The second biodegradation pathway consists of a partial sequence of dechlorination reactions, followed by mineralization of the resulting *cis*DCE or VC. Mineralization (or direct oxidation) is the process in which organic compounds are transformed into inorganic by-products. In mineralization reactions, *cis*DCE and VC serve as electron donors. This combination of partial reductive dechlorination of TCE followed by mineralization of *cis*DCE and VC has been shown to result in complete biodegradation of chlorinated ethene contaminants in many ground-water systems (Wiedemeier and Chappelle,1998).

Reductive dechlorination of TCE by microorganisms requires anaerobic ground-water conditions. TCE serves as an electron acceptor during this process, and a chlorine atom is removed and replaced with a hydrogen atom to form *cis*DCE. All three isomers of dichloroethene can theoretically be produced, but *cis*DCE is by far a more common daughter product than *trans*-1,2-DCE or 1,1-DCE (Bouwer, 1994). Because TCE is reduced during this process, the reaction must be coupled with the oxidation of another compound. Organic compounds such as natural organic matter, fuel hydrocarbons, or various compounds found in landfill leachate are common electron donors that are oxidized during the reduction of TCE. Reductive dechlorination of TCE has been demonstrated under all anaerobic conditions, but it is reported to be most efficient under sulfate-reducing (Gossett and Zinder, 1996) or methanogenic conditions (McCarty and Semprini, 1994). Reductive dechlorination can also degrade *cis*DCE to VC to ethene, but the sequence can be (and often is) interrupted due to a change to less reducing redox conditions as contaminants are transported downgradient by ground-water flow. TCE is the chlorinated ethene most susceptible to reductive dechlorination, followed by *cis*DCE and VC.

Mineralization of *cis*DCE, DCE or VC by microorganisms requires aerobic or mildly anaerobic ground-water conditions. The chlorinated VOC's serve as electron donors during this process and are degraded to carbon dioxide, water, and chloride. Because *cis*DCE, DCE or VC are oxidized in this process, the reactions must be coupled with the reduction of another compound. Oxygen, manganese (IV), or iron (III) are common electron acceptors that can be reduced during mineralization of *cis*DCE, DCE and VC (Bradley and others, 1998; Bradley and Chapelle, 1998; Bradley and Chapelle, 1997).

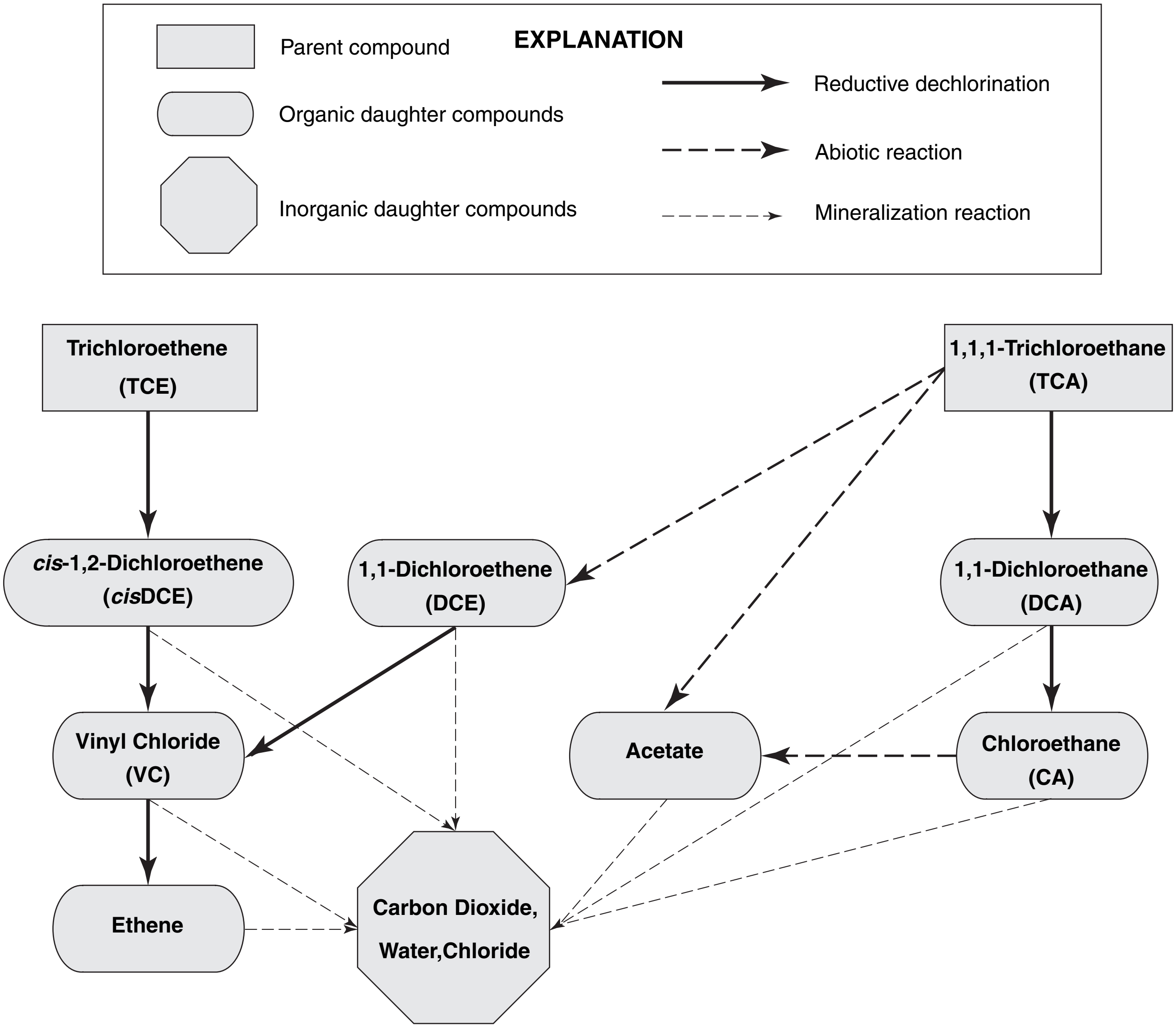

Figure 4. Possible degradation pathways for trichloroethene (TCE) and 1,1,1-trichloroethane (TCA). (Modified from McCarty and Semprini, 1994; after Vogel and McCarty, 1987.)

Mineralization of *cis*DCE, DCE and VC is most efficient under aerobic conditions, is moderately efficient under iron- or manganese-reducing conditions, and is negligible under sulfate-reducing and methanogenic conditions. The rates of *cis*DCE and DCE mineralization are usually somewhat less than those for VC mineralization, and TCE is not readily mineralized.

Given the above processes, the complete degradation of TCE into carbon dioxide, water, and chloride is most effective in aquifers that are sulfate reducing or methanogenic near the TCE source area, and that become aerobic or manganese/iron reducing downgradient of the source area (Chapelle, 1996). Such redox conditions allow reductive dechlorination of TCE and possibly *cis*DCE, followed by mineralization of the resulting *cis*DCE and VC. In order for an aquifer to have the ideal sequence of redox conditions, a substantial source of organic carbon is needed to create strongly reducing conditions near the TCE source area, and a depletion of that organic carbon combined with additional DO (usually from ground-water recharge) is needed downgradient.

1,1,1-Trichloroethane and Other Chlorinated Ethanes

In contrast to TCE, the chlorinated ethane 1,1,1-TCA can be degraded at significant rates by both abiotic and biotic reactions. Abiotic degradation of TCA can occur through two different pathways (fig. 4). Reported half-lives for the abiotic degradation of TCA into acetate range from 0.5–1.7 years, and half-lives for abiotic degradation of TCA into DCE range from 0.8–2.5 years (Vogel and McCarty, 1987). Those half-lives reflect conditions at a temperature of 20 degrees Celsius (°C). Expected field half-lives for TCA at 15°C, a representative ground-water temperature for Area 6, would be about twice as long (1.6 to 5 years) based on data from Jeffers and others (1989). Abiotic degradation of TCA is essentially independent of redox conditions or pH, so it occurs throughout most aquifers. Abiotic degradation of 1,1-DCA and 1,2-DCA is not an important natural attenuation process (Jeffers and others, 1989)—reported half-lives are about 60 to 70 times greater than those for TCA. However, abiotic degradation of CA is very rapid, with a reported half-life of only 0.12 years at 20°C (Bouwer and McCarty, 1983).

Reductive dechlorination of chlorinated ethanes is similar to that for chlorinated ethenes. TCA can be reductively dechlorinated to 1,1-DCA and then to CA (fig. 4). The process occurs under anaerobic conditions only, and is most effective under the more strongly reducing conditions of sulfate reduction and methanogenesis. TCA is more susceptible to reductive dechlorination than are DCA or CA. Less is known about mineralization of chlorinated ethanes. Mineralization of 1,2-DCA under aerobic conditions has been demonstrated previously (Klecka and others, 1998), and mineralization of 1,2-DCA under anaerobic conditions was demonstrated during this investigation. Mineralization of 1,1-DCA has been inferred but not well studied, and mineralization of CA is suspected to be much slower than abiotic degradation of that compound.

Similar to the degradation of TCE, the complete degradation of TCA into carbon dioxide, water, and chloride is also most efficient in aquifers that are strongly reducing near the source area and aerobic or mildly reducing downgradient of the source area. In such a setting, TCA, and possibly DCA, would be reductively dechlorinated in the strongly reducing zone, followed by abiotic degradation of CA and mineralization of any remaining DCA. However, the potential for substantial abiotic degradation of TCA suggests that even in an aerobic aquifer, TCA can degrade into acetate and 1,1-DCE, which could subsequently be degraded into innocuous daughter products. Given that degradation of 1,1-DCE (a very hazardous compound) could possibly be a rate-limiting step in the process, a fully aerobic aquifer system would probably not be as conducive to TCA degradation as would an aquifer system that is anaerobic at the source area and is aerobic downgradient.

Nondestructive Attenuation Processes

Nondestructive natural attenuation processes include those that result only in reducing contaminant concentration, but not the total contaminant mass in a system. The most important non-destructive attenuation processes for chlorinated VOC's in ground water are dispersion, sorption, and volatilization. At Area 6, volatilization of contaminants from ground water is not particularly relevant because the relatively thick (greater than 100 ft) unsaturated zone greatly restricts vapor transport from ground water to the atmosphere.

Dispersion causes a contaminant plume to spread out in directions that are longitudinal and transverse to the direction of ground-water flow. Dispersion is relevant to natural attenuation because it dilutes contaminant concentrations by spreading the contaminant into a larger volume of aquifer. Dispersion is attributed to both mechanical and molecular diffusion processes, but mechanical dispersion is the dominant process at normal ground-water flow velocities (Davis and others, 1993). The more heterogeneous the materials of an aquifer are, the greater the dispersion will be.

Sorption is the partitioning of dissolved solutes from the ground water onto sediment particles. Sorption is relevant to natural attenuation because it retards the movement of a contaminant plume. Sorption is largely a reversible process so contaminants may not be permanently removed from solution. Sorption is a complex process than includes several mechanisms, but sorption of chlorinated VOC's is primarily dependent on the amount of organic carbon present in the aquifer materials and the hydrophobicity of the contaminant.

The amount of sorption is often quantified by a distribution coefficient (K_d). A K_d for a given contaminant in a given material is defined as the ratio of the sorbed contaminant concentration to the dissolved contaminant concentration. Two methods for determining K_d are commonly used—an empirical method based on measurements of total organic carbon (TOC) in an aquifer and laboratory tests conducted with native aquifer materials. The first method relies on batch and column test results from numerous studies for a range of contaminant types and concentrations and aquifer conditions. Because K_d's were found to be well correlated with the fraction of organic carbon in aquifer materials (Schwarzenbach and Westall, 1981), a simple estimation formula was developed.

$$K_d = K_{oc} \times TOC \quad ,$$

where K_{oc} is a compound specific coefficient obtained from previous laboratory studies, and TOC is the total organic carbon fraction measured in the aquifer material. The second method for determining K_d involves performing laboratory batch and column tests on aquifer samples. Details required for this procedure can be found in Weidemeier and others (1998).

Once a K_d is determined, the equilibrium distribution of contaminants between the sorbed and dissolved phases generally can be described by a linear sorption isotherm.

$$C_S = K_d \times C_W \quad ,$$

where C_S is the amount of contaminant sorbed to the solid phase, and C_W is the amount in solution. Although such an equilibrium between contaminant concentrations in the sorbed and solution phases is often assumed, numerous experiments have shown that desorption can be rate-limiting, and previously sorbed contaminants can be slowly released into solution over a period of weeks or months (Harmon and others, 1989).

METHODS AND DATA ANALYSIS

The framework for evaluating natural attenuation at Area 6 is summarized below. Methods used to sample and analyze ground water and to conduct microcosm experiments are also described.

Framework for Evaluation of Natural Attenuation

This evaluation of natural attenuation was based on existing guidelines and protocols, particularly from Wiedemeier and Chapelle (1998), Weidemeier and others (1998), Weidemeier and others (1996), and U.S. Environmental Protection Agency (1997). The focus was on evaluating the effectiveness of natural attenuation alone without the superimposed effects of the existing pump and treat remediation system. The following evidence for natural attenuation was evaluated:

1. Historical ground-water contaminant data were examined to ascertain any clear trends of declining contaminant concentrations or mass over time.
2. Contaminant, geochemical, and hydrogeologic data were examined for evidence of active natural attenuation processes. In particular, redox conditions conducive to different degradation reactions were identified and mapped, the presence and distribution of degradation daughter products were evaluated, and hydrogeologic data were used in an analytical model employed to estimate field contaminant degradation rates.
3. Laboratory microcosm studies were completed to directly demonstrate that selected degradation processes (aerobic and anaerobic mineralization of VC and DCA) could be mediated by indigenous microorganisms under ambient ground-water conditions at Area 6.

After the evidence was compiled, an analytical ground-water flow and transport model was used to determine if natural attenuation could meet site remediation goals if the existing pump and treat system was not operating. When natural attenuation was found to not be effective at meeting remediation goals throughout Area 6, the analytical model was used to estimate target contaminant concentrations that pump and treat must attain before natural attenuation alone would be an effective alternative.

Data evaluated in this investigation included contaminant chemistry and hydrogeological data collected by the Navy and the City of Oak Harbor between 1989 and 1998, and ground-water geochemistry data collected by the USGS in 1997 and 1998. The laboratory microcosm experiments were conducted by the USGS in 1998 using aquifer sediments collected by the USGS from Area 6.

Ground-water Sampling and Analyses

All ground-water samples were collected from monitoring or extraction wells at Area 6 and at the Oak Harbor landfill (fig. 2). Methods and analyses used by the Navy for collecting and analyzing contaminant chemistry ground-water samples are described by URS Consultants (1993b), IT Corporation (1997), and Foster Wheeler Environmental (1997). In general, those samples were collected with either bottom-filling bailers (before 1993) or submersible pumps after purging at least three well volumes of water from the wells, and after allowing ground-water temperature, pH, specific conductance, and turbidity measurements to stabilize to within 10 percent. Samples were analyzed for VOC's using EPA Method 601. The City of Oak Harbor used similar sampling procedures, but with EPA Method SW 8010 for VOC analyses.

Ground-water samples were collected by USGS from 38 wells during April 1997, from 19 wells during September 1997, and from 32 wells during June 1998. Those samples were collected with a submersible bladder pump after purging at least three well volumes of water from the wells, and allowing ground-water temperature, pH, specific conductance, and DO measured in a flow-through chamber to stabilize to within ten percent. Wells sampled and analytes measured during each of the three USGS sampling rounds are indicated in table 1.

Water temperature, specific conductance, pH, and DO were measured in the field immediately after collection of unfiltered samples. Specific conductance and temperature were measured using a temperature compensated probe and meter that were checked daily with standard reference solutions. The pH was measured using a temperature-compensated meter calibrated daily with two pH standards. Dissolved oxygen was usually measured with a temperature-compensated meter that was calibrated daily with water-saturated air. Dissolved oxygen measurements were also made for many samples using 0–1 mg/L CHEMettes Rhodazine-D^{TM} colorimetric ampules, and (or) a modified Winkler titration (Radtke and others, 1998). Those analyses were done to confirm or substitute for meter readings, particularly when DO concentrations were less than 1 mg/L. Alkalinity was measured by field titration of a sample that was passed through a 0.45-micrometer filter.

Concentration of nitrate, manganese, iron (II), total iron, and hydrogen sulfide were measured in the field immediately after collection and filtration of samples through a 0.45-micrometer membrane filter. Analyses were made using a Hach Model 2010 spectrophotometer, according to Hach analytical method numbers 8039, 8034, 8146, 8008, and 8131 (Hach Company, 1998). Selected samples also were filtered into polyethylene bottles, acidified with nitric acid to pH less than 2, chilled, and sent to the USGS National Water Quality Laboratory (NWQL) for measurement of manganese and iron using inductively coupled plasma (ICP) spectroscopy. Those lab analyses were done to check the accuracy of the field analyses, and the data were comparable (table 2). Samples for sulfate and chloride analyses were filtered into polyethylene bottles with no preservatives added, and analyte concentrations were measured using ion chromatography at the NWQL.

Total organic carbon (TOC) was measured in the USGS laboratory in Ocala, Fla. using EPA Method 415.1. The unfiltered TOC samples were collected in amber glass bottles and chilled to less than 4°C prior to shipment to the laboratory. Dissolved methane samples were collected in serum bottles and sealed with teflon septa. The samples were preserved with mercuric chloride, chilled, and analyzed using gas chromatography (Kampbell and others, 1989) at the USGS Water Resources Division District laboratory at Columbia, S.C.

Table 1. Wells sampled and analytes measured by the U.S. Geological Survey at Area 6 during each of three sampling rounds

[X, indicates that it was measured for the specified well during the sampling round; 1, April 1997; 2, September 1997; 3, June, 1998]

Well identification number	Dissolved oxygen			Total organic carbon			Nitrate			Manganese			Iron (II)			Total iron			Hydrogen sulfide			Sulfate		
	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3
PW-1	X	–	X	–	–	X	X	–	X	–	–	X	X	–	X	X	–	X	X	–	X	–	–	X
PW-2	X	–	–	–	–	–	X	–	–	–	–	–	X	–	–	X	–	–	X	–	–	–	–	–
PW-3	X	–	X	–	–	X	X	–	X	–	–	X	X	–	X	X	–	X	X	–	X	–	–	X
PW-4	X	–	X	–	–	X	X	–	X	–	–	X	X	–	X	X	–	X	X	–	X	–	–	X
PW-5	X	–	X	–	–	X	X	–	X	–	–	X	X	–	X	X	–	X	X	–	X	–	–	X
PW-6	X	–	X	–	–	X	X	–	X	–	–	X	X	–	X	X	–	X	X	–	X	–	–	X
PW-7	X	–	X	–	–	X	X	–	X	–	–	X	X	–	X	X	–	X	X	–	X	–	–	X
PW-8	X	–	X	–	–	X	X	–	X	–	–	X	X	–	X	X	–	X	X	–	X	–	–	X
PW-9	X	–	X	–	–	X	X	–	X	–	–	X	X	–	X	X	–	X	X	–	X	–	–	X
6-S-2	X	X	–	–	X	–	X	X	–	–	X	–	X	X	–	–	X	–	X	X	–	–	X	–
6-S-3	X	–	X	–	–	X	X	–	X	–	–	X	X	–	X	–	–	X	X	–	X	–	–	X
6-S-4	–	X	X	–	X	X	–	X	X	–	X	X	–	X	X	–	X	X	–	X	X	–	X	X
6-S-6	X	X	–	–	X	–	X	X	–	–	X	–	X	X	–	X	X	–	X	X	–	–	X	–
6-S-7	X	–	–	–	–	–	X	–	–	–	–	–	X	–	–	X	–	–	X	–	–	–	–	–
6-S-10	X	–	X	–	–	X	X	–	X	–	–	X	X	–	X	X	–	X	X	–	X	–	–	X
6-S-11	X	–	X	–	–	X	X	–	X	–	–	X	X	–	X	X	–	X	X	–	X	–	–	X
6-S-12	X	–	X	–	–	X	X	–	X	–	–	X	X	–	X	X	–	X	X	–	X	–	–	X
6-S-13	X	X	X	–	X	X	X	X	X	–	X	X	X	X	X	X	X	X	X	X	X	–	X	X
6-S-14	–	X	X	–	X	X	–	–	X	–	X	X	–	X	X	–	X	X	–	X	X	–	X	X
6-S-19	X	X	X	–	X	X	X	X	X	–	X	X	X	X	X	–	X	X	X	X	X	–	X	X
6-S-21	X	–	X	–	–	X	X	–	X	–	–	X	X	–	X	X	–	X	X	–	X	–	–	X
6-S-22	X	X	X	–	X	X	X	X	X	–	X	X	X	X	X	X	X	X	X	X	X	–	X	X
6-S-23	–	X	X	–	X	X	–	X	X	–	X	X	–	X	X	–	X	X	–	X	X	–	X	X
6-S-24	X	–	X	–	–	X	X	–	X	–	–	X	X	–	X	X	–	X	X	–	X	–	–	X
6-S-25	X	X	X	–	X	X	X	–	X	–	X	X	X	X	X	X	X	X	X	X	X	–	X	X
6-S-26	X	X	–	–	X	–	X	X	–	–	X	–	X	X	–	X	X	–	X	X	–	–	X	–
6-S-27	X	X	X	–	X	X	X	–	X	–	X	X	X	X	X	X	X	X	X	X	X	–	X	X
6-S-28	X	X	X	–	X	X	X	X	X	–	X	X	X	X	X	X	X	X	X	X	X	–	X	X
6-S-29	X	X	X	–	X	X	X	X	X	–	X	X	X	X	X	X	X	X	X	X	X	–	X	X
MW-3B	–	–	X	–	–	X	–	–	X	–	–	X	–	–	X	–	–	X	–	–	X	–	–	X
MW-6	X	–	X	–	–	X	X	–	X	–	–	X	X	–	X	X	–	X	X	–	X	–	–	X
MW-7	X	X	–	–	X	–	X	X	–	–	X	–	X	X	–	X	X	–	X	X	–	–	X	–
MW-8	X	–	X	–	–	X	X	–	X	–	–	X	X	–	X	–	–	X	X	–	X	–	–	X
MW-9	X	–	X	–	–	X	X	–	X	–	–	X	X	–	X	–	–	X	X	–	X	–	–	X
MW-10	–	X	X	–	X	X	–	X	X	–	X	X	–	X	X	–	X	X	–	X	X	–	X	X
MW-11	X	–	–	–	–	–	X	–	–	–	–	–	X	–	–	X	–	–	X	–	–	–	–	–
MW-14	X	–	–	–	–	–	X	–	–	–	–	–	X	–	–	X	–	–	X	–	–	–	–	–
N6-37	X	X	X	–	X	–	X	X	X	–	X	X	X	X	X	X	X	X	–	X	X	–	X	X
N6-38	X	X	X	–	X	–	X	X	X	–	X	X	–	X	X	–	X	X	X	X	X	–	X	X
6-|-1	X	–	–	–	–	–	X	–	–	–	–	–	X	–	–	X	–	–	X	–	–	–	–	–
6-|-3	X	X	–	–	X	–	X	X	–	–	X	–	X	X	–	X	X	–	X	X	–	–	X	–
6-|-6	X	–	–	–	–	–	X	–	–	–	–	–	X	–	–	X	–	–	X	–	–	–	–	–
6-|-8	X	–	–	–	–	–	X	–	–	–	–	–	X	–	–	X	–	–	X	–	–	–	–	–
6-D-3	–	X	–	–	X	–	–	X	–	–	X	–	–	X	–	–	X	–	–	X	–	–	X	–
P-6	–	–	X	–	–	X	–	–	X	–	–	X	–	–	X	–	–	X	–	–	X	–	–	X

Table 1. Wells sampled and analytes measured by the U.S. Geological Survey at Area 6 during each of three sampling rounds—*Continued*

[X, indicates that it was measured for the specified well during the sampling round; 1, April 1997; 2, September 1997; 3, June, 1998]

Well identification number	Dissolved hydrogen			Alkalinity			Chloride			pH			Specific conductance			Temperature			Methane		
	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3	1	2	3
PW-1	–	–	–	–	–	X	–	–	X	X	–	X	X	–	X	–	–	X	–	–	–
PW-2	–	–	–	–	–	–	–	–	–	X	–	–	X	–	–	–	–	–	–	–	–
PW-3	–	–	–	–	–	X	–	–	X	X	–	X	X	–	X	–	–	X	–	–	–
PW-4	–	–	–	–	–	X	–	–	X	X	–	X	X	–	X	–	–	X	X	–	–
PW-5	–	–	–	–	–	X	–	–	X	X	–	X	X	–	X	–	–	X	–	–	–
PW-6	–	–	–	–	–	X	–	–	X	X	–	X	X	–	X	–	–	X	X	–	–
PW-7	–	–	–	–	–	X	–	–	X	X	–	X	X	–	X	–	–	X	X	–	–
PW-8	–	–	–	–	–	X	–	–	X	–	–	X	–	–	X	–	–	X	X	–	–
PW-9	–	–	–	–	–	X	–	–	X	X	–	X	X	–	X	–	–	X	–	–	–
6-S-2	–	–	–	–	–	–	–	X	–	X	–	–	X	X	–	–	–	–	–	–	--
6-S-3	–	–	–	–	–	X	–	–	X	X	–	X	X	–	X	–	–	X	–	–	--
6-S-4	–	–	X	–	–	X	–	X	X	–	–	X	–	X	X	–	–	X	–	–	--
6-S-6	–	–	–	–	–	–	–	X	–	X	–	–	X	X	–	–	–	–	–	–	--
6-S-7	–	–	–	–	–	–	–	–	–	X	–	–	X	–	–	–	–	–	X	–	--
6-S-10	–	–	X	–	–	X	–	–	X	X	–	X	X	–	X	–	–	X	X	–	--
6-S-11	–	–	X	–	–	X	–	–	X	X	–	X	X	–	X	–	–	X	X	–	--
6-S-12	–	–	X	–	–	X	–	–	X	X	–	X	X	–	X	–	–	X	X	–	--
6-S-13	–	–	X	–	–	X	–	X	X	X	–	X	X	X	X	–	–	X	–	–	--
6-S-14	–	–	X	–	–	X	–	X	X	–	–	X	–	X	X	–	–	X	–	–	--
6-S-19	–	–	X	–	–	X	–	X	X	X	–	X	X	X	X	–	–	X	–	–	--
6-S-21	–	–	–	–	–	X	–	–	X	X	–	X	X	–	X	–	–	X	–	–	--
6-S-22	–	–	X	–	–	X	–	X	X	X	–	X	X	X	X	–	–	X	–	–	--
6-S-23	–	–	X	–	–	X	–	X	X	–	–	X	–	X	X	–	–	X	–	–	--
6-S-24	–	–	–	–	–	X	–	–	X	X	–	X	X	–	X	–	–	X	–	–	--
6-S-25	–	–	–	–	–	X	–	X	X	X	–	X	X	X	X	–	–	X	–	–	--
6-S-26	–	–	–	–	–	–	–	X	–	X	–	–	X	X	–	–	–	–	–	–	--
6-S-27	–	–	–	–	–	X	–	X	X	X	–	X	X	X	X	–	–	X	–	–	--
6-S-28	–	–	X	–	–	X	–	X	X	X	–	X	X	X	X	–	–	X	–	–	--
6-S-29	–	–	X	–	–	X	–	X	X	X	–	X	X	X	X	–	–	X	–	–	--
MW-3B	–	–	–	–	–	X	–	–	X	–	–	X	–	–	X	–	–	X	–	–	–
MW-6	–	–	X	–	–	X	–	–	X	X	–	X	X	–	X	–	–	X	–	–	–
MW-7	–	–	–	–	–	–	–	X	–	X	–	–	X	X	–	–	–	–	–	–	–
MW-8	–	–	X	–	–	X	–	–	X	X	–	X	X	–	X	–	–	X	X	–	–
MW-9	–	–	X	–	–	X	–	–	X	X	–	X	X	–	X	–	–	X	–	–	–
MW-10	–	–	X	–	–	X	–	X	X	–	–	X	–	X	X	–	–	X	–	–	–
MW-11	–	–	–	–	–	–	–	–	–	X	–	–	X	–	–	–	–	–	–	–	–
MW-14	–	–	–	–	–	–	–	–	–	X	–	–	X	–	–	–	–	–	–	–	–
N6-37	–	–	–	–	–	X	–	X	X	X	–	X	X	X	X	–	–	X	–	–	–
N6-38	–	–	–	–	–	X	–	X	X	X	–	X	X	X	X	–	–	X	–	–	–
6-I-1	–	–	–	–	–	–	–	–	–	X	–	–	X	–	–	–	–	–	–	–	–
6-I-3	–	–	–	–	–	–	–	X	–	X	–	–	X	X	–	–	–	–	X	–	–
6-I-6	–	–	–	–	–	–	–	–	–	X	–	–	X	–	–	–	–	–	X	–	–
6-I-8	–	–	–	–	–	–	–	–	–	X	–	–	X	–	–	–	–	–	X	–	–
6-D-3	–	–	–	–	–	–	–	X	–	–	–	–	–	X	–	–	–	–	–	–	–
P-6	–	–	–	–	–	X	–	–	X	–	–	X	–	–	X	–	–	X	–	–	–

Hydrogen in ground water was sampled using the bubble-strip method of Chapelle and others (1997) and measured using a field gas chromatograph equipped with a reduction gas detector. Gas samples were first collected and analyzed after 20 minutes of stripping; subsequent samples were collected and analyzed at about 5-minute intervals until measured hydrogen concentrations stabilized to within 10 percent.

Table 2. Comparison of field and laboratory measurements of total dissolved iron and manganese concentrations in ground-water samples from Area 6, Naval Air Station Whidbey Island, 1998

[**mg/L**, milligrams per liter]

Well identification number	Field measured manganese (mg/L)	Laboratory measured manganese (mg/L)	Field measured iron (mg/L)	Laboratory measured iron (mg/L)
PW-1	0.016	0.007	0.01	<0.01
PW-3	0.007	0.052	0.002	<0.01
PW-4	2.1	2	0.99	0.87
PW-5	0.088	0.046	0.61	0.25
PW-6	0.16	0.14	0.39	0.13
PW-7	4.0	4.1	4.0	3.8
PW-8	1.5	1.3	0.59	0.43
PW-9	0.6	0.59	7.1	9.4
6-S-3	0.028	<0.012	0.06	0.035
6-S-4	2.9	2.9	0.19	0.15
6-S-10	0.063	0.057	0.01	<0.01
6-S-11	0.36	0.32	8.1	9.1
6-S-12	0.53	0.6	15	17
6-S-13	0.62	0.57	0.29	0.27
6-S-14	0.57	0.5	2.4	2.1
6-S-19	0.71	0.65	1.1	0.93
6-S-21	0.1	0.097	1.2	1.1
6-S-22	0.031	0.016	0.1	<0.01
6-S-23	0.17	0.15	0.3	0.25
6-S-24	0.043	<0.004	0.02	<0.01
6-S-25	0.01	0.05	0.01	<0.01
6-S-27	0.043	<0.004	0.01	<0.01
6-S-28	0.12	0.44	3.3	3.8
6-S-29	1.1	1.1	1.6	1.5
MW-3B	0.014	<0.004	0.02	<0.01
MW-6	0.22	0.21	1.6	1.6
MW-8	0.024	0.012	0.02	<0.01
MW-9	8.6	7.1	3.03	3.3
MW-10	2.2	3	18	16
N6-37	0.021	0.009	0.01	<0.01
N6-38	0.01	<0.004	0.01	<0.01
P-6	0.13	0.083	0.25	0.21

Microcosm Experiments

Microcosm experiments were conducted to test if microorganisms indigenous to the shallow aquifer at Area 6 could readily mineralize VC and DCA to carbon dioxide under aerobic or anaerobic conditions. Sediments from near the top of the saturated zone of the shallow aquifer at well 6-S-29 were used to construct microcosms for VC experiments. That well is in the southern contaminant plume where VC mineralization is most relevant. Sediments from near the top of the saturated zone of the shallow aquifer at piezometer P-8 (near well PW-3) were used to construct microcosms for DCA experiments. That well is in the western contaminant plume where DCA mineralization is most relevant. The aquifer sediment samples were collected with a clean, flame-sterilized split-spoon sampler during drilling of the respective wells in April 1997 and May 1998. After collection, samples were immediately transferred to cleaned and baked glass jars, sealed, and stored at 4°C prior to initiation of the experiments. The experiments were conducted in the USGS Water Resources Division District laboratory in Columbia, S.C. Microcosm preparation, incubation conditions, and analytical procedures have been described previously (Bradley and Chapelle, 1996; Bradley and Chapelle, 1997; Bradley and others, 1998; and Bradley and Chapelle, 1998) and are summarized below.

The VC experiment involved adding carbon-14 uniformly radio-labeled VC ([1,2 ^{14}C] VC) to sealed vials containing moist aquifer sediments, and monitoring the amount of $^{14}CO_2$ formed at subsequent time points to determine if significant amounts of VC had been mineralized to CO_2. Two test conditions (aerobic and iron-reducing) and a sterilized abiotic control were set up, and six observations over time were made of each treatment. Observations consisted of triplicate measurements of the ^{14}C present in the microcosm as CO_2. An initial $^{14}CO_2$ measurement was made immediately following the addition of the radio-labeled VC, and the sixth and final observation was 27 days later.

The VC microcosms were prepared using 10 grams (g) of moist aquifer sediment that was aseptically added to 20-milliliter (mL) serum vials. The aerobic microcosms were amended with 1 ml of sterile distilled water while anaerobic microcosms were amended with 1 mL of Fe-ethylenediamineteraacetic acid (Fe-EDTA) solution. The Fe-EDTA was added to ensure that iron-reducing conditions prevailed throughout the entire experiment in the anaerobic microcosms. The headspaces of the aerobic microcosms were unmodified air while the headspaces in anaerobic microcosms were flushed with 100 percent helium prior to sealing. Microcosms were sealed with thick butyl rubber stoppers that were modified to suspend a base trap assembly in the headspace above the saturated sediment. Sterilized abiotic control microcosms were made following identical procedures except that prior to the addition of the radio labeled VC, the microcosms were autoclaved at 15 pounds per square inch (lb/in^2) and 121°C for 1 hour.

After the microcosms had been sealed, 10 microliters (µL) of radio-labeled VC (approximately 50,000 disintegrations per minute (dpm)) were added to each microcosm by injection through the rubber stopper directly into the sediment to minimize volatilization. The radio-labeled VC was obtained from New England Nuclear with a reported purity of 97.7 percent. The microcosms were incubated in the dark at approximately 20°C. Triplicate measurements of $^{14}CO_2$ were made at the outset of the experiment and at five subsequent time points. At each time point, the microcosms were acidified with 0.4 mL of HPO_4, followed by the addition of 0.3 mL of 3N KOH to the base trap assembly to collect CO_2. Typically, 24 hours were allowed for CO_2 in the microcosm headspace to accumulate in the base trap. The CO_2 was then removed from the base trap and the amount of $^{14}CO_2$ was quantified by scintillation counting. Twenty-four hour base-trap recovery efficiency was checked using 20,000- and 50,000-dpm sodium bicarbonate. The radiometric detector was calibrated by liquid scintillation counting using $H^{14}CO_3$.

The DCA experiment involved adding carbon-14 uniformly radio-labeled DCA ([1,2-^{14}C] DCA) to sealed vials containing moist aquifer sediments, and monitoring the amount of $^{14}CO_2$ and other possible degradation products formed at subsequent time points. The isomer 1,2-DCA was used as a model DCA compound, rather than the 1,1-DCA isomer that is more prevalent at Area 6, because radio-labeled 1,1-DCA was not readily available from chemical supply companies. Slight differences in degradation potential of the different DCA isomers may exist, but those differences were not assessed in this experiment. Three test conditions (aerobic, anaerobic/unamended, and anaerobic/MnO_2 amended) and a sterilized abiotic control were set up, and three observations over time were made of each.

The DCA microcosms were prepared using 15 g of moist aquifer sediment that was aseptically added to 20-mL serum vials. The aerobic and anaerobic/unamended microcosms were amended with 1 mL of anoxic, sterile distilled water, while the anaerobic/amended microcosms were amended with 1 mL (approximately 0.05 g) of an anoxic sterile slurry of poorly crystalline MnO_2 to assure manganese(IV)-reducing conditions. The headspace of the aerobic microcosms was unmodified air while the headspace in anaerobic microcosms was flushed with 100-percent helium prior to sealing. Microcosms were sealed with thick butyl rubber stoppers. Sterilized abiotic control microcosms were made following identical procedures except that prior to the addition of the radio labeled DCA, the microcosms were autoclaved twice at 15 lb/in^2 at 121°C for 1 hour.

The microcosms were pre-incubated for 5 days at 20°C and then amended with approximately 0.05 microcuries (*u*Ci) of [1,2-^{14}C] DCA (specific activity = 11.1 millicuries per millimole) by injection through the rubber stopper directly into the sediment to minimize volatilization. The radio-labeled DCA obtained from Sigma Chemical Company had a reported purity of greater than 98 percent. Initial dissolved 1,2-DCA concentrations in equilibrium with the headspace were estimated, based on experimentally determined adsorption and Henry's law coefficients, to be 90 micrograms per liter (µg/L). The microcosms were incubated in the dark at approximately 20°C. Headspace concentrations of potential degradation products ^{14}C-chloroethane, ^{14}C-ethene, $^{14}CH_4$, and $^{14}CO_2$ were monitored periodically by removing 0.5 mL of headspace and analyzing by gas chromatography using a thermal conductivity detector coupled in series to a radiometric detector. The removed sample volume was replaced with helium.

CONCEPTUAL MODEL OF NATURAL ATTENUATION AT AREA 6

There are substantial differences in natural attenuation processes that are occurring in the two contaminant plumes in the shallow aquifer beneath the site. Natural attenuation has not been particularly effective at controlling the migration of contaminants in the western contaminant plume, the narrow north to south trending TCA and TCE plume that originates near the former hazardous waste storage area. That is because the predominantly aerobic redox conditions in the western plume are not favorable for reductive dechlorination of TCA and TCE. In contrast, natural attenuation has been effective both at controlling the migration and reducing the mass of contaminants in the southern contaminant plume, the plume of predominately VC and TCA that originates beneath the former Navy landfill and spreads southward across the southern site boundary. That is because the strongly anaerobic redox conditions beneath the Area 6 landfill are favorable for reductive dechlorination of TCA, TCE, and their dichlorinated degradation products, and the mildly reducing and aerobic redox conditions downgradient of the landfill are favorable for subsequent mineralization of the resulting VC.

The ground-water flow path for the western plume begins just north of Ault Field Road. Ground water recharged through the organic-rich wetlands at that location is quickly depleted of oxygen due to microbial oxidation of naturally occurring organic matter. Between Ault Field Road and the contaminant source area, oxygenated recharge from nonwetland areas creates aerobic conditions in the upper part of the aquifer, but drives anaerobic ground water downward to the lower part of the aquifer. Residual TCA and TCE beneath the source area are flushed into the ground water by oxygenated recharge and are transported southward. The low organic carbon content of sediments and ground water in the upper part of the shallow aquifer allow aerobic conditions to persist downgradient to the southwest site boundary. The aerobic conditions inhibit reductive dechlorination of TCE and TCA, although abiotic degradation of TCA occurs. The anaerobic conditions in the lower part of the shallow aquifer persist downgradient due to the higher organic carbon content of the finer-grained sediments deep in the aquifer. The anaerobic conditions at depth allow some reductive dechlorination of TCE and TCA, but most of the contamination in the western plume migrates southward in the upper part of the aquifer. The pump and treat system does not appear to be affecting redox conditions in the western plume, but the extraction wells are stopping or substantially reducing plume expansion.

The ground-water flow path for the southern plume begins just northeast of Area 6 in a non-wetland area where the oxygenated recharge results in aerobic ground water. Ground water throughout the shallow aquifer quickly becomes anaerobic as it flows beneath the former Navy landfill and mixes with landfill leachate enriched with organic carbon. Within and beneath the landfill, strongly reducing conditions promote substantial reductive dechlorination of TCA and TCE. The reductive dechlorination intermediates DCA and *cis*DCE further degrade to CA and VC in the strongly reducing zone. VC is slowest to degrade in the strongly reducing ground water beneath the landfill, so it persists downgradient of the landfill. As the organic carbon in landfill leachate is consumed, and as oxygenated precipitation recharges ground water south of the Area 6 landfill, the redox conditions in the relatively thin saturated zone become less and less reducing in the direction of ground-water flow. The mildly reducing and aerobic conditions downgradient of Area 6 landfill lead to mineralization of VC, CA, and possibly any remaining DCA or *cis*DCE. Thus, the possibility is low that those contaminants are being transported in detectable concentrations more than a few hundred feet beyond the southern site boundary. The small amount of TCA observed downgradient of the Area 6 landfill is not biodegraded in the mildly reducing or aerobic ground water, but it does degrade abiotically to form DCE. DCE is not likely detectable downgradient of the site boundary because there is little TCA available to degrade, and because any DCE that is formed is subsequently mineralized.

The nondestructive natural attenuation processes of volatilization, sorption, and dispersion are not of great significance at Area 6. Volatilization of contaminants from ground water is minimal due to the thick unsaturated zone. Sorption of contaminants to aquifer materials is minimal due to the sandy nature and low organic carbon content of the aquifer sediments. Hydrodynamic dispersion likely has some impact on contaminant concentrations, but the relatively homogeneous aquifer materials at Area 6 are not conducive to substantial dispersion of a contaminant plume.

SITE CHARACTERIZATION RELATED TO NATURAL ATTENUATION

The following sections describe the hydrogeology of the shallow aquifer, redox conditions in ground water, and the distribution of VOC's in ground water at Area 6. This information forms the basis for the subsequent evaluation of natural attenuation.

Hydrogeology of the Shallow Aquifer

The shallow aquifer beneath Area 6 is contained within the Vashon glacial advance outwash sediments. The saturated portion of the aquifer is composed of a fine to medium sand that gradually becomes finer and siltier with depth. Thin layers of gravel are present in the unit. The organic carbon content of aquifer materials is low, ranging from 0.03 to 0.1 percent by weight. The saturated thickness of the aquifer ranges from about 40 ft along the western site boundary to about 15 ft along the eastern site boundary. The aquifer is generally unconfined, with water levels ranging from about 20 to 145 ft below the ground surface (75 to 95 ft above sea level).

Ground-water flow in the shallow aquifer is generally north to south, and the overall flow direction has not been substantially altered by the operation of the pump and treat remediation system (figs. 5-6). A divide between northward and southward flowing ground water exists somewhere in the wetland area just north Ault Field Road where the water table is only about 5 ft below the land surface. The average horizontal hydraulic gradient in January 1998 was about 0.005 ft/ft, nearly double that observed in January 1995. The change in gradient was likely due to operation of the pump and treat system (with infiltration of treated water in a swale running from near well MW-15 to past well 6-S-26) combined with more ground-water recharge resulting from increased precipitation during 1996–98. Pumping tests conducted in 1998 at wells PW-1, PW-3, PW-5, and PW-9 yielded an average horizontal hydraulic conductivity estimate of 87 feet per day (ft/d) (Foster Wheeler Environmental, 1998b), with a range of 47 to 126 ft/d. Those estimates compared well with previous estimates from slug tests and single well aquifer tests (URS Consultants, 1993b). Using the average hydraulic gradient and conductivity estimates, and assuming an aquifer porosity of 25 percent, ground water flow velocity beneath Area 6 was estimated to be about 320 and 640 ft/yr in 1995 and 1998, respectively.

The vertical hydraulic gradient between the shallow aquifer to the next deepest aquifer is positive downward at about 0.17 to 0.35 ft/ft, but the vertical hydraulic gradient at paired wells within the shallow aquifer itself is variable both in direction and magnitude across the site.

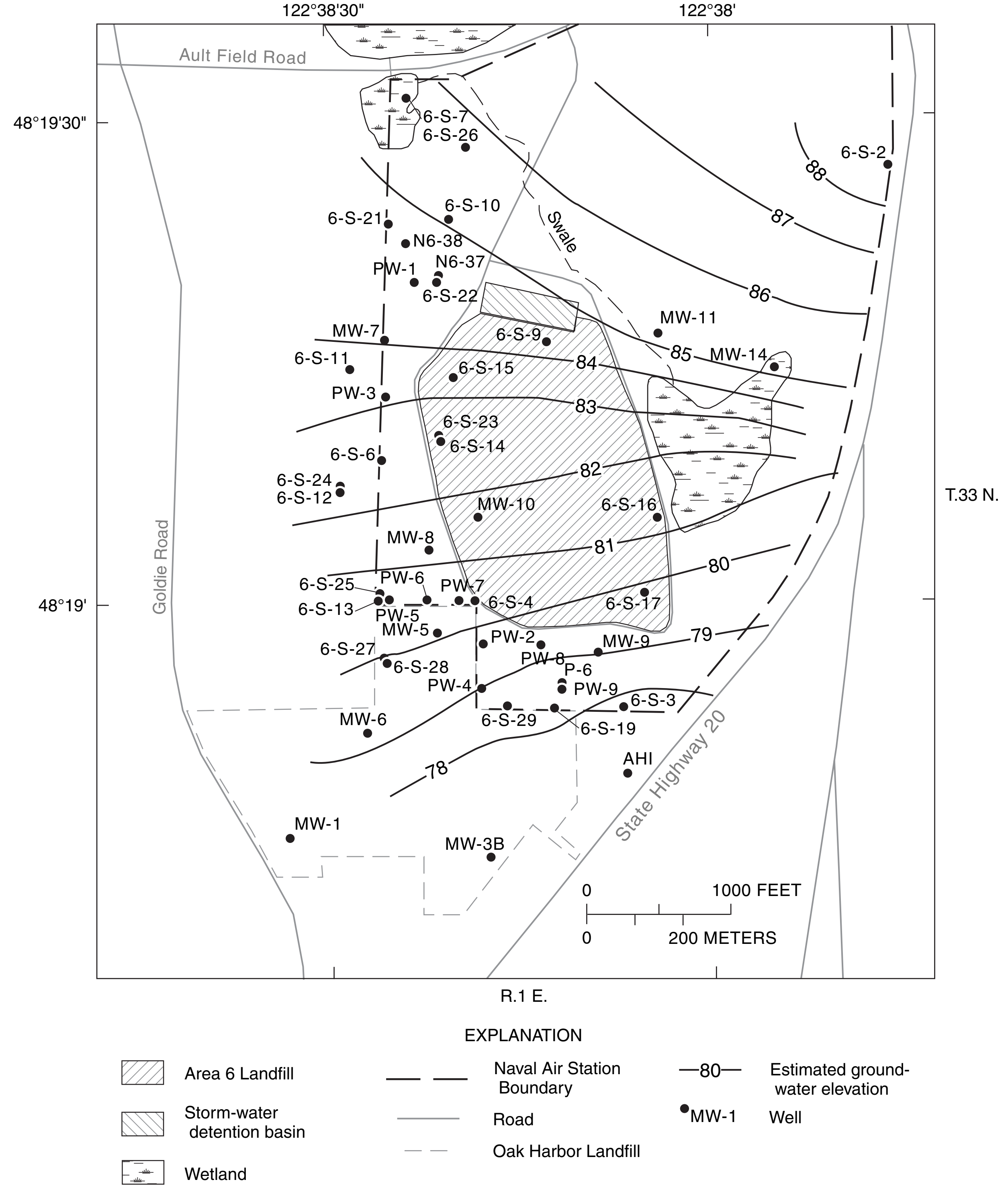

Figure 5. Elevation of ground-water surface in the shallow aquifer beneath Area 6, January 1995. Elevations are in feet above mean sea-level. (From IT Corporation, 1995.)

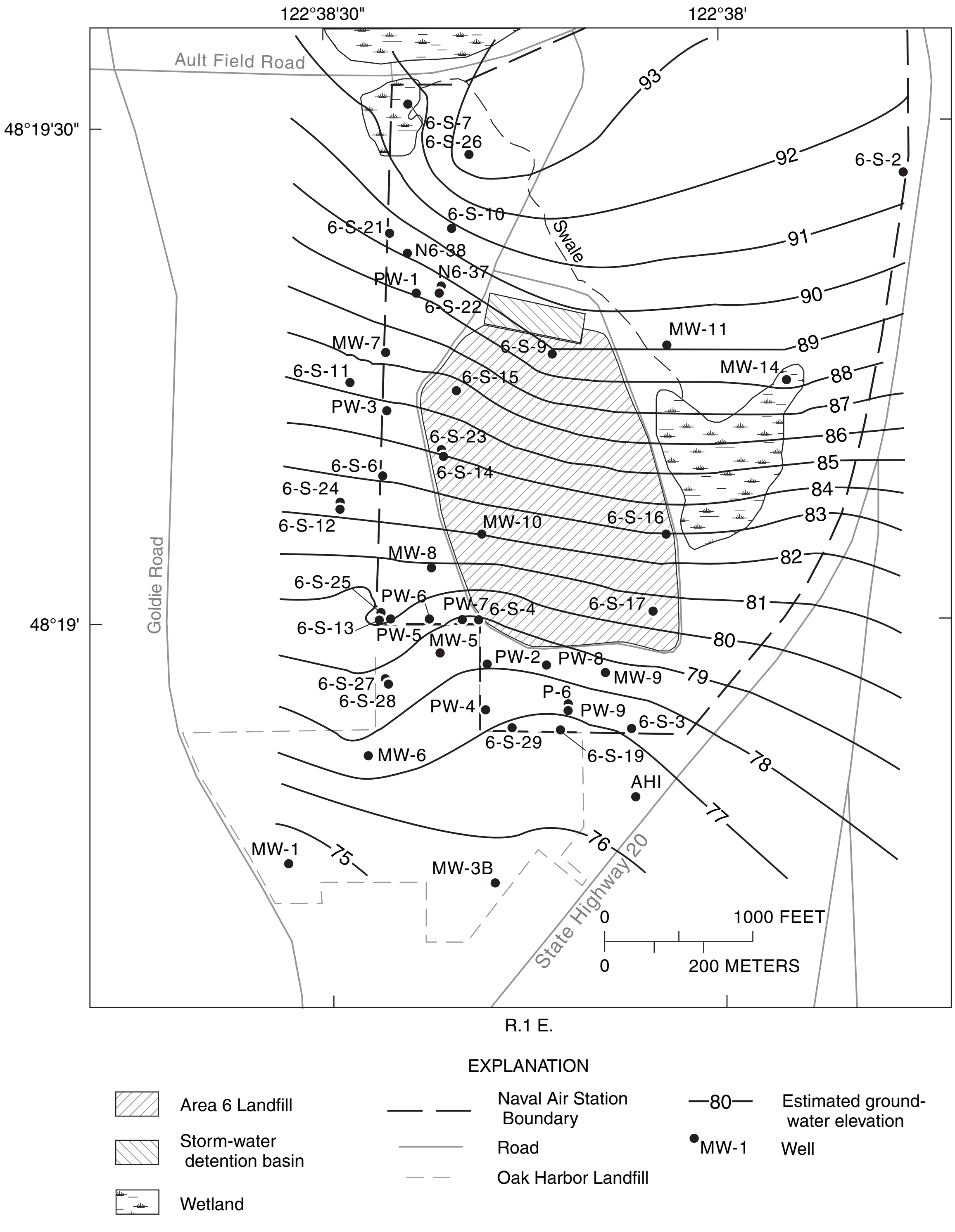

Figure 6. Elevation of ground-water surface in the shallow aquifer beneath Area 6, January 1998. Elevations are in feet above mean sea-level. (From Foster Wheeler Environmental, 1998c.)

From north to south across the site, the gradient is upward at about 0.01 ft/ft between wells N6-37 and 6-S-22, is downward at about 0.0009 ft/ft between wells 6-S-23 and 6-S-14, is upward again at about 0.001 ft/ft between wells 6-S-24 and 6-S-12, is downward again at about 0.006 ft/ft between wells 6-S-25 and 6-S-13, and is downward still at 0.0009 ft/ft at wells 6-S-27 and 6-S-28. Those directions and magnitudes have changed little between 1991 and 1998.

Sorption does not likely have a substantial effect on contaminant transport in the shallow aquifer due to relatively low K_{oc} values of the contaminants and low organic carbon contents of the aquifer materials. Calculated distribution coefficients for the contaminants of concern at Area 6 ranged from 0.003 to 0.15, resulting in estimated contaminant transport retardation factors of 1.2 to 2.2 (URS Consultants, 1993b). Thus, the contaminants are transported in ground water at rates ranging from about 45 percent to near 85 percent of the rate at which ground water is moving.

Dispersion does not likely have a substantial effect on contaminant transport in the shallow aquifer primarily due to the relatively homogeneous aquifer materials and high transmissivities. The long, narrow shape of the western contaminant plume that existed even before pump and treat began is evidence of minimal lateral dispersion in the aquifer.

Redox Conditions in Ground Water

The redox conditions of ground water at Area 6 were determined primarily from the distribution of DO, manganese (Mn(II)), iron (II), sulfate, methane, and dissolved hydrogen that were measured during the June 1998 sampling period (table 3). Supplemental data from the April and September 1997 sampling periods were used for wells not sampled during 1998. The redox conditions identified during June 1998 were generally the same as those during the two previous less extensive sampling periods. Nitrate was analyzed in selected samples, but the data were too few to provide insight into redox conditions. Hydrogen sulfide was analyzed in all samples, but none of the concentrations exceeded the estimated detection limit of 0.01 mg/L for the field analytical method, so those data also provided little insight. There are few historical data available to characterize past redox conditions for the site, so only current conditions were characterized. The June 1998 samples (with the exception of those from the extraction wells) were collected during a period when the pump and treat system was not operating, so those data are somewhat representative of conditions without the imposed effects of pump and treat.

Other geochemical parameters measured that provide insight into redox-related processes include TOC, alkalinity, and pH. High concentrations of TOC indicate that the organic carbon needed to drive reductive dechlorination reactions is available. Alkalinity, which increases in nearly direct proportion to the amount of carbon dioxide being produced by microbial oxidation of organic compounds, is a measure of the overall microbial activity in ground water. Alkalinity can be affected by inorganic chemical reactions, but is mostly affected by microbial activity in the calcium-carbonate poor glacial sediments at Area 6. The pH can affect the presence and activity of microbial populations. Microbes capable of degrading chlorinated VOC's generally prefer pH values between 6 to 8, and all pH measurements at Area 6 fell within that range.

The areal distribution of aerobic, mildly reducing, and strongly reducing redox zones was delineated for both the upper part and the lower part of the shallow aquifer. There were enough data to identify a clear difference in redox conditions between the upper and the lower parts of the shallow aquifer in the western plume, where the saturated thickness of the aquifer is about 40 ft. Fewer data were available to identify redox differences between the upper and lower parts of the shallow aquifer in the southern plume, where the saturated thickness of the aquifer ranges from 30 to only 15 ft.

Redox Conditions in the Upper Part of the Shallow Aquifer

Ground water is aerobic in the upper part of the shallow aquifer in the western contaminant plume and in the area downgradient of the southern contaminant plume (fig. 7). Ground water with measured DO concentration greater than 1 mg/L was considered to be aerobic. The aerobic zone was delineated using data from shallow wells (those screened in the top portion of the aquifer) and from mixed wells (those screened across the entire aquifer). It was assumed that if DO exceeded 1 mg/L in mixed wells, then the upper part of the aquifer in that location was aerobic. That assumption was made because DO data from shallow/deep well pairs and from multiple sample depths in wells MW-7, PW-5, and PW-9 (table 3) showed that the upper part of the aquifer consistently had more DO than the lower part.

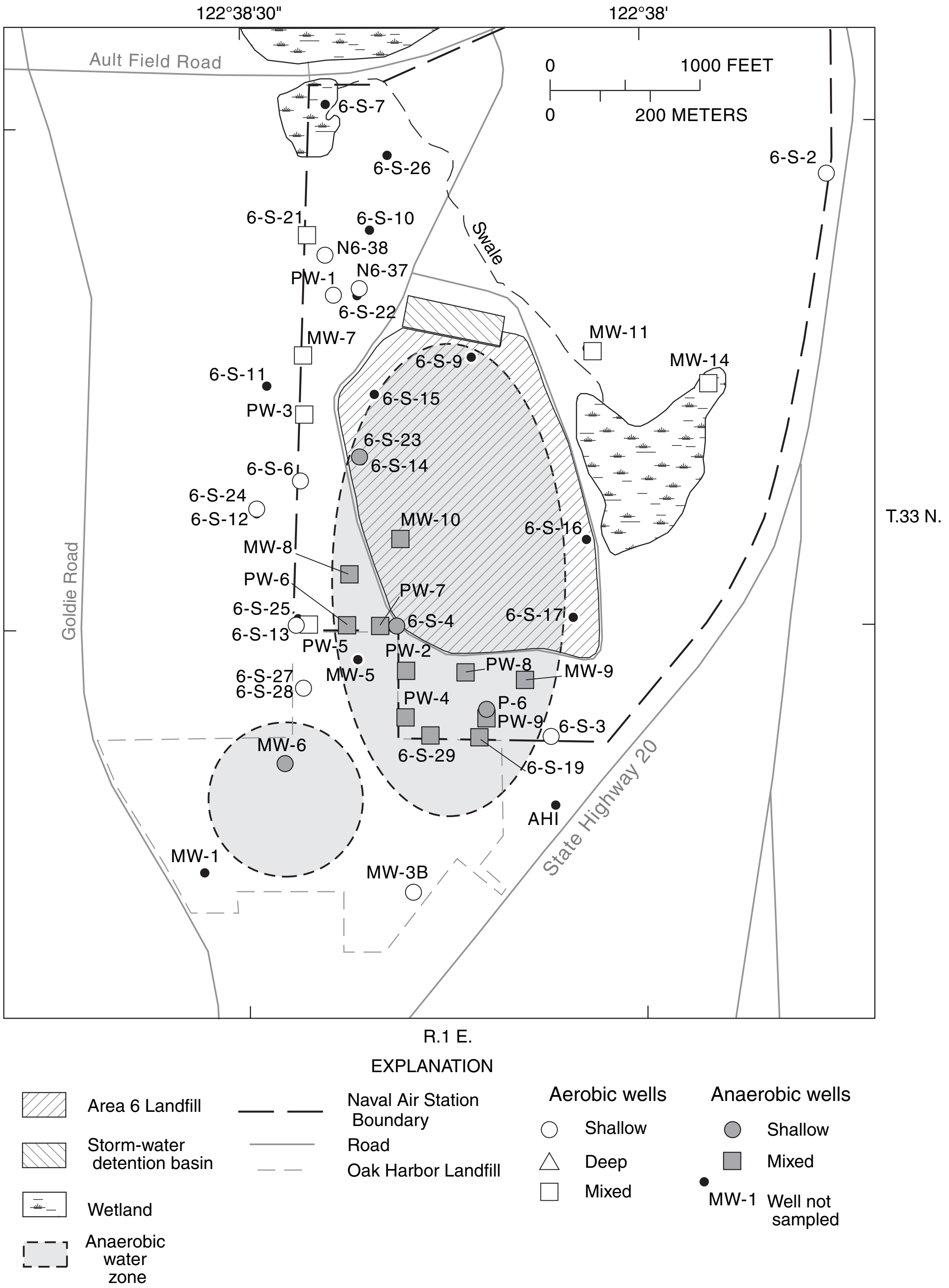

Figure 7. Zones of anaerobic ground water (dissolved oxygen less than 1 milligram per liter) in the upper part of the shallow aquifer at Area 6.

Table 3. Redox sampling data at Area 6, Naval Air Station Whidbey Island, analyzed by U.S. Geological Survey, June 1998

[Supplemental data from April and September, 1997 used for several wells. **H_2**, dissolved hydrogen: **mg/L**, milligrams per liter; **$CaCO_3$**, calcium carbonate; **nmol/L**, nanomoles per liter; **µS/cm**, microsiemens per centimeter at 25 degrees Celsius; –, not analyzed; <, less than; R, replicate analyses]

Well identification number	Dissolved oxygen (mg/L)	Dissolved H_2 (nmol/L)	Total organic carbon (mg/L)	Nitrate (mg/L)	Manganese (mg/L)	Iron (II) (mg/L)	Sulfide (mg/L)	Sulfate (mg/L)
Upper part of aquifer, aerobic wells								
Western plume wells, from north to south								
6-S-21	2.7	–	1.5	–	0.097	0.09	<0.01	6.6
N6-38	10	–	1.2	–	<0.004	0.01	<0.01	<0.1
N6-37	3.4	–	1.9	–	0.009	0.00	<0.01	11
PW-1	2.3	–	2.1	–	0.007	0.00	<0.01	13
MW-7								
at 116 feet	5.0	–	2.3	13	<0.004	0.00	<0.01	20
at 126 feet	1.2	–	–	–	0.031	0.00	<0.01.	8.6
at 136 feet	1.8	–	1.4	21	0.11	0.01	<0.01	20
PW-3	2.2	–	1.4	–	0.052	0.00	<0.01	14
6-S-6	1.2	–	1.1	–	0.20	0.01	<0.01	11
6-S-24	7.6	–	0.6	–	<0.004	0.00	<0.01	33
6-S-25	6.7	–	0.3	–	0.005	0.00	<0.01	14
PW-5	1.8	–	1.6	–	0.047	0.52	<0.01	17
at 118 feet	0.7	–	1.1	–	0.030	0.02	<0.01	17
at 128 feet	0.3	–	1.5	–	0.045	0.02	<0.01	18
at 138 feet	0.2	–	1.3	–	0.073	0.00	<0.01	17
6-S-27	5.7	–	0.7	–	<0.004	0.01	<0.01	26
Wells located east and south of landfill								
6-S-2	8.4	–	0.8	–	0.20	0.02	<0.01	12
MW-11	3.8	–	–	–	–	0.00	<0.01	–
MW-14	7.2	–	–	–	–	0.00	<0.01	–
6-S-3	2.5	–	1.4	–	<0.012	0.04	<0.01	10
MW-3B	4.6	–	0.5	–	<0.004	0.00	<0.01	68
Upper part of aquifer, anaerobic wells								
Southern plume wells, from north to south								
6-S-23	0.3	0.7	2.9	0.04	0.15	0.29	<0.01	31
MW-10	0.0	1.5	15	–	3.0	16	<0.01	3.0
MW-10(R)	0.0	–	–	–	3.2	15	<0.01	3.4
MW-8	0.2	0.2	2.1	–	0.012	0.00	<0.01	21
PW-6	0.5	–	1.9	–	0.14	0.13	<0.01	18
PW-7	0.0	–	15	–	4.1	2.7	<0.01	4.1
6-S-4	0.2	0.2	8.9	–	2.9	0.15	<0.01	3.4
PW-2	0.8	–	–	–	–	4.6	<0.01	–

Table 3. Redox sampling data at Area 6, Naval Air Station Whidbey Island, analyzed by U.S. Geological Survey, June 1998

[Supplemental data from April and September, 1997 used for several wells. **H_2**, dissolved hydrogen: **mg/L**, milligrams per liter; **$CaCO_3$**, calcium carbonate; **nmol/L**, nanomoles per liter; **µS/cm**, microsiemens per centimeter at 25 degrees Celsius; –, not analyzed; <, less than; R, replicate analyses]

Well identification number	Dissolved methane (mg/L)	Alkalinity (mg/L as $CaCO_3$)	Chloride (mg/L)	pH	Specific conductance (µS/cm)	Total dissolved solids (mg/L)	Water temperature (degrees Celsius)	Sampling date
Upper part of aquifer, aerobic wells								
Western plume wells, from north to south								
6-S-21	–	125	8.0	7.1	279	175	10.4	June 1998
N6-38	–	111	9.3	6.9	231	–	11.3	June 1998
N6-37	–	301	11	6.3	632	355	15.8	June 1998
PW-1	–	247	12	6.9	520	300	12.0	June 1998
MW-7								
at 116 feet	–	256	13	6.6	843	422	11.3	June 1998
at 126 feet	–	185	8.3	6.7	471	242	11.9	June 1998
at 136 feet	–	232	14	6.8	677	411	11.7	June 1998
PW-3	–	356	12	7.0	699	405	11.7	June 1998
6-S-6	–	–	8.5	7.1	978	–	–	September 1997
6-S-24	–	142	41	7.7	490	275	11.1	June 1998
6-S-25	–	194	22	7.9	469	267	11.1	June 1998
PW-5	–	300	31	7.5	700	393	11.3	June 1998
at 118 feet	–	340	26	7.1	737	423	11.7	June 1998
at 128 feet	–	342	28	7.1	744	423	11.7	June 1998
at 138 feet	–	332	28	7.2	751	424	11.3	June 1998
6-S-27	–	143	26	7.9	426	245	11.4	June 1998
Wells located east and south of landfill								
6-S-2	–	–	18	7.3	330	–	–	September 1997
MW-11	–	–	–	7.8	1,090	–	–	April 1997
MW-14	–	–	–	7.4	551	–	–	April 1997
6-S-3	–	867	23	6.9	1,480	870	11.6	June 1998
MW-3B	–	288	45	7.7	808	466	17.7	June 1998
Upper part of aquifer, anaerobic wells								
Southern plume wells, from north to south								
6-S-23	–	711	16	6.7	1,320	763	13.4	June 1998
MW-10	–	731	24	6.9	1,320	733	12.5	June 1998
MW-10(R)	–	733	24	6.8	1,320	755	13.1	June 1988
MW-8	0.7	731	12	6.8	1,300	736	11.6	June 1998
PW-6	<0.01	683	12	7.0	1,230	713	11.5	June 1998
PW-7	11	1,590	45	7.0	1,990	1170	13.4	June 1998
6-S-4	–	1,240	53	6.8	2,160	1260	11.5	June 1998
PW-2	–	–	–	6.9	690	–	–	April 1997
PW-8	3.3	1,480	55	6.9	2,430	1450	12.0	June 1998

Table 3. Redox sampling data at Area 6, Naval Air Station Whidbey Island, analyzed by U.S. Geological Survey, June 1998

[Supplemental data from April and September, 1997 used for several wells. **H_2**, dissolved hydrogen: **mg/L**, milligrams per liter; **$CaCO_3$**, calcium carbonate; **nmol/L**, nanomoles per liter; **µS/cm**, microsiemens per centimeter at 25 degrees Celsius; –, not analyzed; <, less than; R, replicate analyses]

Well identification number	Dissolved oxygen (mg/L)	Dissolved H_2 (nmol/L)	Total organic carbon (mg/L)	Nitrate (mg/L)	Manganese (mg/L)	Iron (II) (mg/L)	Sulfide (mg/L)	Sulfate (mg/L)
PW-8	0.1	–	7.4	–	1.3	0.43	<0.01	7.2
MW-9	0.0	–	32	–	7.1	2.9	<0.01	2.5
PW-9	0.2	–	3.8	–	0.59	5.3	<0.01	11
at 131 feet	0.5	–	–	–	0.14	1.2	<0.01	7.9
at 136 feet	0.1	–	5.5	–	0.32	5.2	<0.01	8.7
at 139 feet	0.0	–	4.8	–	0.28	3.0	<0.01	8.5
P-6	0.1	–	5.7	–	0.083	0.22	<0.01	5.8
PW-4	0.7	–	5.5	–	2.0	0.83	<0.01	12
6-S-19	0.1	0.8	2.6	–	0.65	1.1	<0.01	16
6-S-29	0.0	0.6	6.8	–	1.1	1.6	<0.01	6.4
MW-6	0.0	0.8	11	–	0.21	1.3	<0.01	60
mW-6(R)	0.0	–	–	–	0.29	1.3	<0.01	62
Lower part of aquifer, anaerobic wells								
Western plume wells, from north to south								
6-S-7	0.0	–	–	–	–	20	–	–
6-S-26	0.0	–	1.8	–	0.1	0.81	<0.01	9.2
6-S-10	0.3	1.5	1.9	–	0.057	0.01	<0.01	6.5
6-S-22	0.0	0.8	1.4	–	0.016	0.01	<0.01	11
6-S-11	0.0	0.9	8.1	–	0.32	7.2	<0.01	28
6-S-14	0.0	1.3	1.2	0.14	0.50	2.3	<0.01	23
6-S-12	0.0	0.4	6.5	–	0.60	12.5	<0.01	30
6-S-13	0.2	0.9	3.0	0.02	0.57	0.25	<0.01	18
6-S-28	0.0	0.8	6.0	–	0.44	3.2	<0.01	26

Table 3. Redox sampling data at Area 6, Naval Air Station Whidbey Island, analyzed by U.S. Geological Survey, June 1998

[Supplemental data from April and September, 1997 used for several wells. **H_2**, dissolved hydrogen: **mg/L**, milligrams per liter; **$CaCO_3$**, calcium carbonate; **nmol/L**, nanomoles per liter; **µS/cm**, microsiemens per centimeter at 25 degrees Celsius; –, not analyzed; <, less than; R, replicate analyses]

Well identification number	Dissolved methane (mg/L)	Alkalinity (mg/L as $CaCO_3$)	Chloride (mg/L)	pH	Specific conductance (µS/cm)	Total dissolved solids (mg/L)	Water temperature (degrees Celsius)	Sampling date
MW-9	–	1,190	39	6.9	2,100	1130	11.2	June 1998
PW-9	–	1,020	80	6.8	1,900	1110	11.8	June 1998
at 131 feet	–	1,160	40	6.8	1,980	1210	13.9	June 1998
at 136 feet	–	1,240	48	6.8	1,990	1280	13.4	June 1998
at 139 feet	–	1,190	43	6.9	1,950	1260	11.2	June 1998
P-6	–	1,240	40	6.9	2,120	1120	12.4	June 1998
PW-4	9.2	1,080	43	6.9	1,890	1110	12.0	June 1998
6-S-19	–	975	36	6.8	1,710	981	11.7	June 1998
6-S-29	–	1,420	66	6.8	2,600	1410	11.8	June 1998
MW-6	–	181	27	7.5	567	337	15.7	June 1998
NW-6(R)	–	181	28	7.5	548	338	–	June 1998
Lower part of aquifer, anaerobic wells								
Western plume wells, from north to south								
6-S-7	<0.01	264	–	6.9	550	–	–	April 1997
6-S-26	–	–	6.3	7.5	232	–	–	September 1997
6-S-10	<0.01	170	6.2	7.6	352	206	10.4	June 1998
6-S-22	–	294	9.2	6.7	566	322	11.5	June 1998
6-S-11	<0.01	133	21	7.2	404	241	11.3	June 1998
6-S-14	–	430	13	6.9	842	482	12.7	June 1998
6-S-12	<0.01	127	30	7.1	424	252	10.7	June 1998
6-S-13	–	209	50	7.6	573	320	10.5	June 1998
6-S-28	–	164	32	7.2	480	280	11.1	June 1998

Ground water is anaerobic in the upper part of the shallow aquifer beneath and immediately downgradient from the Area 6 landfill (fig. 7). The anaerobic conditions are due to oxygen depletion from microbial degradation of the abundant organic carbon in the Area 6 landfill leachate. The landfill leachate plume can be identified by TOC concentrations greater than about 2 mg/L, specific conductance (SC) greater than about 1,000 microsiemens per centimeter (µS/cm), total dissolved solids (TDS) greater than about 700 mg/L, and alkalinity greater than about 500 mg/L $CaCO_3$ (table 3). The leachate plume corresponds closely to the anaerobic zone in the upper part of the shallow aquifer. The separate area of shallow anaerobic ground water around well MW-6 beneath the Oak Harbor landfill likely is due to organic-carbon rich leachate from the Oak Harbor landfill. Dissolved oxygen was not measured by the City of Oak Harbor in their landfill well MW-1, but they did measure low TOC, iron (II), and manganese concentrations, which together suggest that the ground water near MW-1 is aerobic.

Redox conditions within the shallow anaerobic ground-water zone were further delineated into a strongly reducing zone and a mildly reducing zone (fig. 8). The strongly reducing zone includes most of the ground water directly beneath the Area 6 landfill. Dissolved hydrogen, methane, and sulfate data were used to delineate the strongly reducing zone, where ground waters are sulfate-reducing or methanogenic. The dissolved hydrogen concentration of 1.5 nmol/L in well MW-10 is evidence for sulfate-reducing conditions beneath the landfill. Sulfate concentrations are depleted to below 10 mg/L within and immediately downgradient of the strongly reducing zone, also suggesting that sulfate reduction is occurring. Methane concentrations of up to 11 mg/L in the landfill leachate plume indicate that methanogenesis is occurring somewhere beneath the landfill and dissolved methane is being transported downgradient. The exact extent of the strongly reducing zone could not be confirmed, in part because dissolved hydrogen could not be measured in any of the extraction wells (labeled with a PW prefix in the figures and tables). The steel casings and electric pumps in the extraction wells can generate dissolved hydrogen that is not representative of ambient ground water conditions.

The mildly reducing zone includes the remainder of anaerobic ground-water in the upper part of the shallow aquifer. Ground waters in that zone appear to be a mix of manganese and iron reducing. Dissolved hydrogen concentrations ranged from 0.2 to 0.8 nmol/L, which are indicative of manganese and iron reduction. Dissolved manganese and iron (II) concentrations, which are by-products of manganese and iron reduction, were consistently in the range of milligrams per liter.

Redox Conditions in the Lower Part of the Shallow Aquifer

Ground water is anaerobic in the lower part of the shallow aquifer throughout Area 6 (fig. 9). That anaerobic ground water is due to abundant organic carbon in ground water from landfill leachate. Another probable source of organic carbon is the sediments in the lower part of the shallow aquifer in the western plume. TOC concentrations are elevated in wells 6-S-11, 6-S-12, and 6-S-28, but alkalinity, SC, and TDS values in those wells are much less than values found in landfill leachate-affected wells. Anaerobic ground water was also found in some up-gradient uncontaminated wells (6-S-7 and 6-S-26), due to microbial respiration in organic carbon-rich sediments of the wetland recharge zone located just north of Ault Field Road. Oxygenated recharge to the shallow aquifer south of those wetlands overlies the anaerobic ground water that flows southward in the deeper part of the aquifer.

Redox conditions in the lower part of the shallow aquifer were also delineated into strongly reducing and mildly reducing zones (fig. 9). A northern strongly reducing zone includes the ground water immediately downgradient of the wetland recharge zone near Ault Field Road. Dissolved hydrogen concentration of 1.5 nmol/L in well 6-S-10, and sulfate concentrations of 10 mg/L (less than ambient concentrations upgradient of the well) are evidence for sulfate reducing conditions in that zone. Methane was not detected in the northern strongly reducing zone, so it is unlikely that methanogenesis is also occurring. A southern strongly reducing zone in the lower part of the aquifer was assumed to coincide with the strongly reducing zone in the upper part of the aquifer beneath the Area 6 landfill, although there are few data to vertically delineate redox conditions in that area. Ground waters surrounding the strongly reducing zone in the lower part of the shallow aquifer appear to be iron reducing, with some possible sulfate reduction. Dissolved hydrogen concentrations ranged from 0.4 to 0.9 nmol/L, which are indicative of iron-reduction and a transition to sulfate reduction. Dissolved iron (II) concentrations were consistently measured in the range of milligrams per liter.

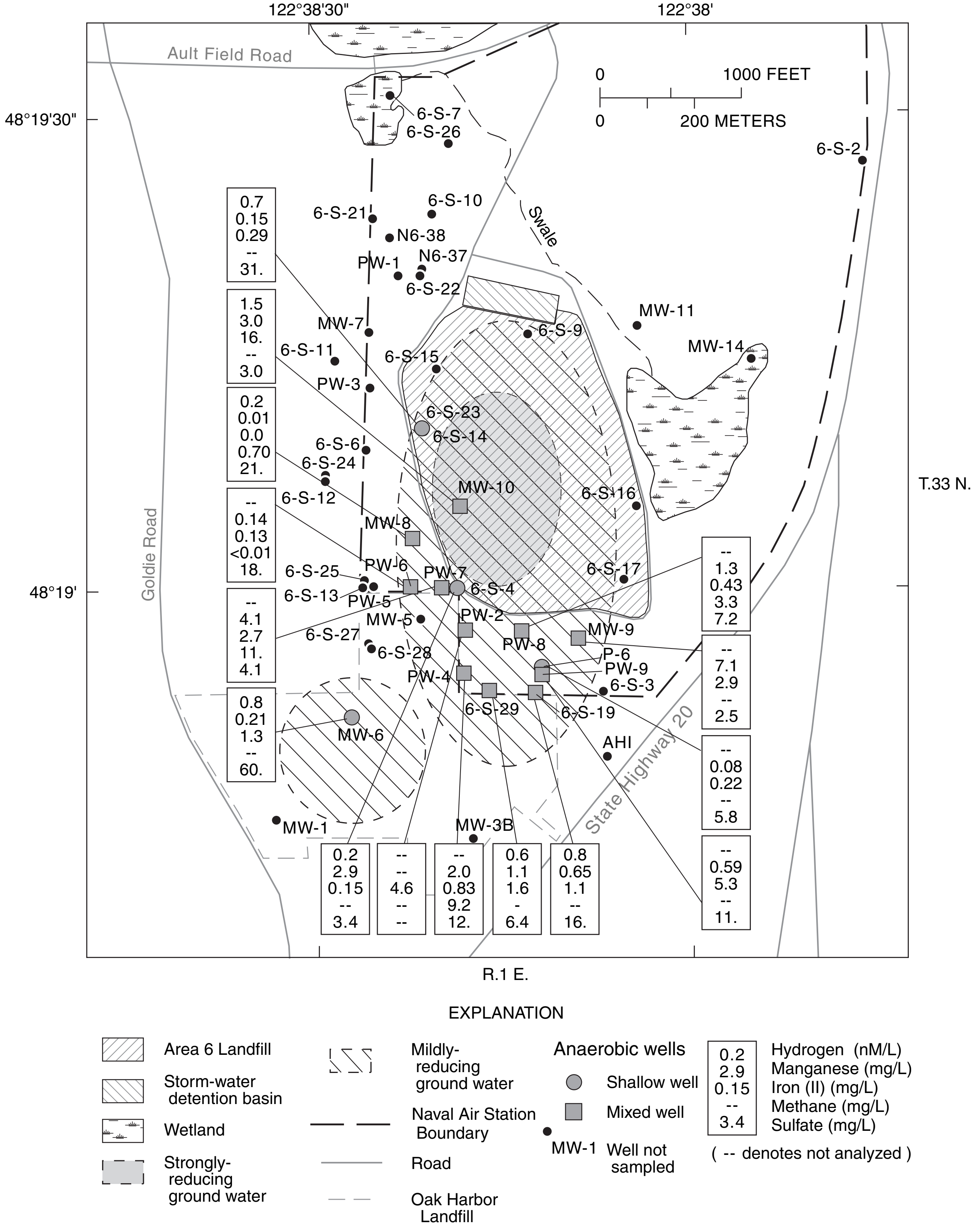

Figure 8. Zones of mildly and strongly reducing ground water in the upper part of the shallow aquifer at Area 6, and related chemistry data.

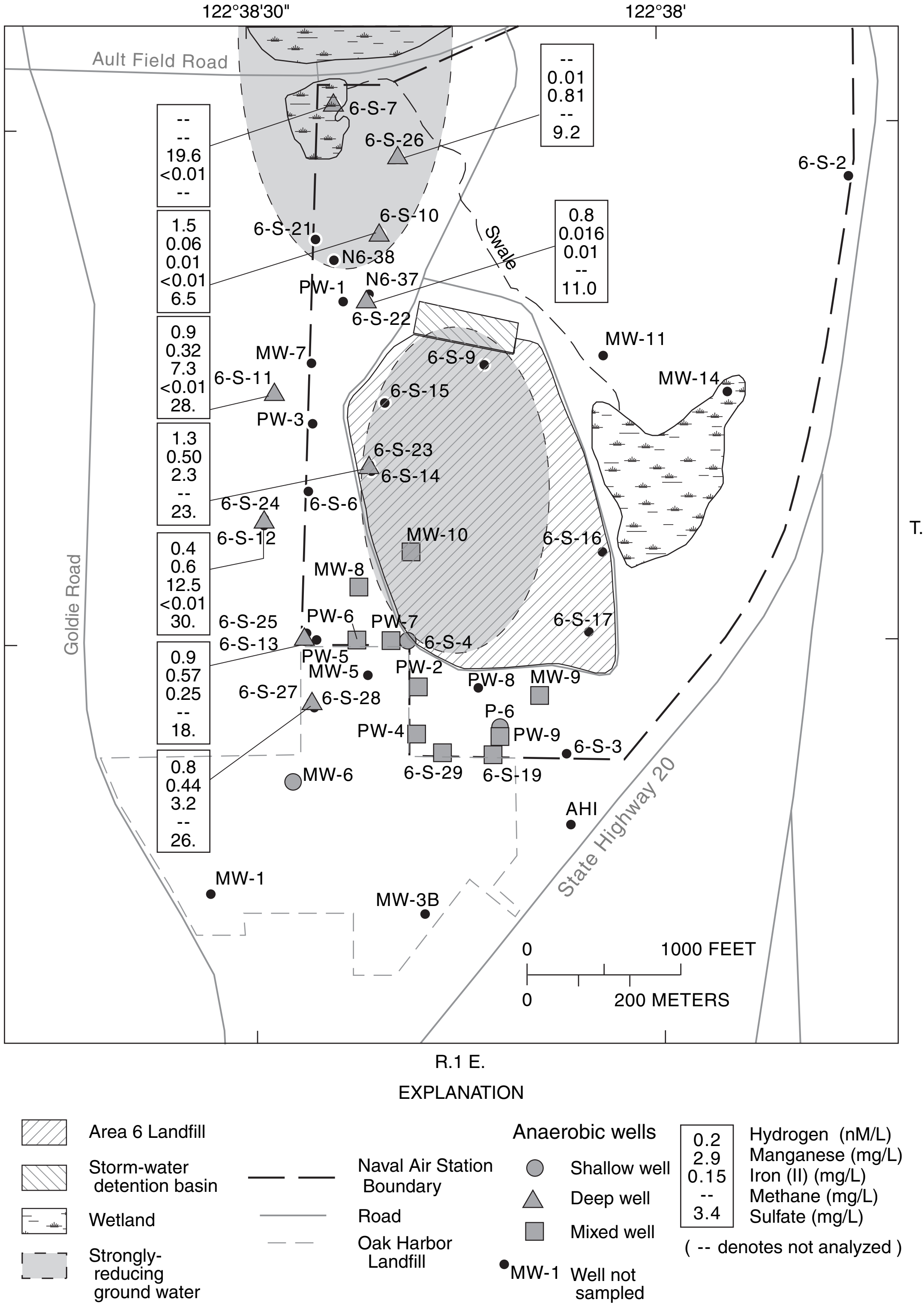

Figure 9. Zones of strongly reducing ground water in the lower part of the shallow aquifer at Area 6, and related chemistry data. Remainder of area is mildly reducing.

Chlorinated Volatile Organic Compounds in Ground Water

The observed 1991, 1995, and 1998 distributions of TCA, TCE, DCA, DCE, *cis*DCE, and VC are shown on figures 10–15 (no data for *cis*DCE were reported for 1995.) The 1991 data are the earliest extensive data available for Area 6, so they represent the first comprehensive picture of the contaminant plumes. The 1995 data were collected just prior to start-up of the pump and treat system, so they represent changes in contaminant concentrations that can be attributed to natural attenuation processes alone. The 1998 data represent subsequent changes due to the combined effects of natural attenuation and active remediation. A complete listing of chlorinated VOC data collected at Area 6 since 1993 is provided in table 4.

The data shown on figures 10–15 are the average concentrations measured during as many as three sampling rounds in 1991, and as many as four sampling rounds in 1998; only one sampling round was completed in 1995 before pump and treat start-up. The data were compiled from the RI report (URS Consultants, 1993b) and from subsequent quarterly technical reports (table 4). In the RI report, discrepancies were found between some data shown in tables, figures, and appendices, so judgement was required to determine which data to use. When averaging data that included a mix of nondetected and detected concentrations, the nondetected concentrations were assumed to equal the detection limit. The data are from wells that are screened at different depths in the shallow aquifer, as identified on the figures. Wells with screens that are 5 to 20 ft long in either the top half or the bottom half of the aquifer are classified as shallow wells or deep wells, respectively. Wells with screens that are 20 ft or longer that include at least two-thirds of the saturated thickness of the aquifer are classified as mixed wells. The estimated concentration contours on the figures are interpretations of the highest concentrations in any 10 ft thick section of the aquifer, so not all shown data were used when estimating those contours.

EVIDENCE FOR NATURAL ATTENUATION

The primary evidence used to evaluate natural attenuation at Area 6 included changes in contaminant concentrations over time, ground-water chemistry data demonstrating contaminant degradation, and results from microcosm studies.

Changes in Contaminant Concentrations over Time

Changes in contaminant concentrations between 1991 and 1995 reflect changes that can be attributed to natural attenuation. Changes in contaminant concentrations between 1995 and 1998 reflect changes that can be attributed to natural attenuation combined with pump and treat and the landfill cap. Analysis of the available data showed that there were not enough monitoring wells or usable analyses, particularly from 1991, to evaluate changes in the mass of contaminants in ground water at the site. Thus, this assessment of contaminant changes over time is based on concentration data alone.

Southern Contaminant Plume

Contaminant concentrations in the southern plume have generally remained unchanged at relatively low values both before and after the onset of active remediation. The parent compound TCE was not detected in the southern plume during 1991–95. It has been detected sporadically during 1995–98, but concentrations exceeded the 5 µg/L cleanup level specified in the ROD (URS Consultants, 1993a) only once (6.6 µg/L in well 6-S-19, March 1996). The parent compound TCA was detected frequently in the southern plume during 1991–95, but the observed concentrations were always at least a factor of 20 less than the 200 µg/L clean-up level. The TCA concentrations remained essentially unchanged during 1995–98. The TCE degradation product *cis*DCE was not detected in the southern plume during 1991, and was not analyzed for in 1995. It has been detected consistently in wells MW-10 and PW-7 at concentrations between 0.2 and 1.9 µg/L during 1997-98; those values are well below the 70 µg/L cleanup level. The degradation product DCA was detected sporadically at up to 21 µg/L between 1991–95, also well below the 800 µg/L cleanup level. Between 1995–98, DCA continued to be detected in some wells at concentrations up to 31 µg/L. The DCA data show a consistent trend of increasing concentration between 1995–98 in the extraction wells PW-7, PW-2, and PW-4.

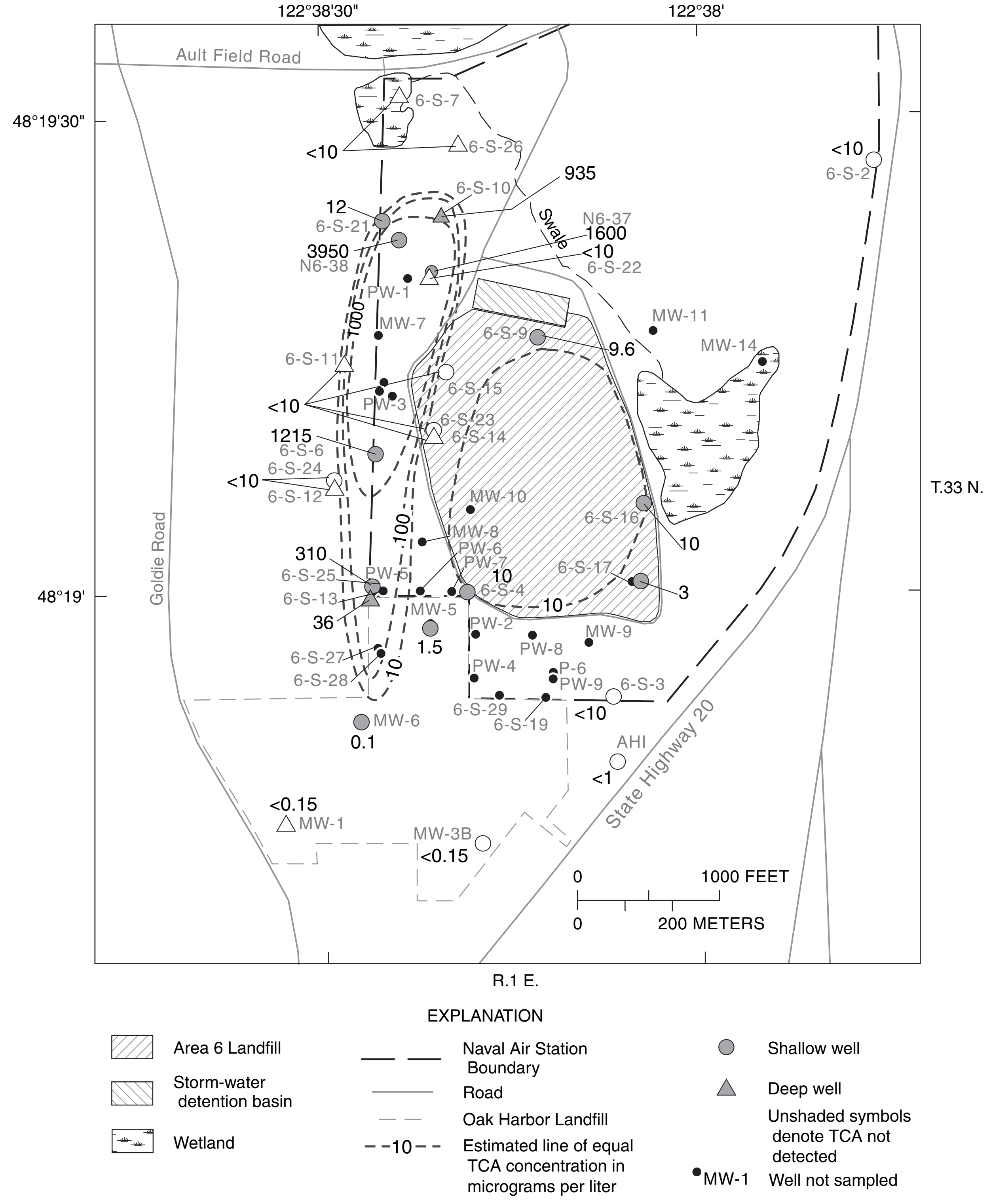

Figure 10a. Concentrations of 1,1,1-trichloroethane (TCA) in shallow ground water at Area 6, 1991. Units are micrograms per liter.

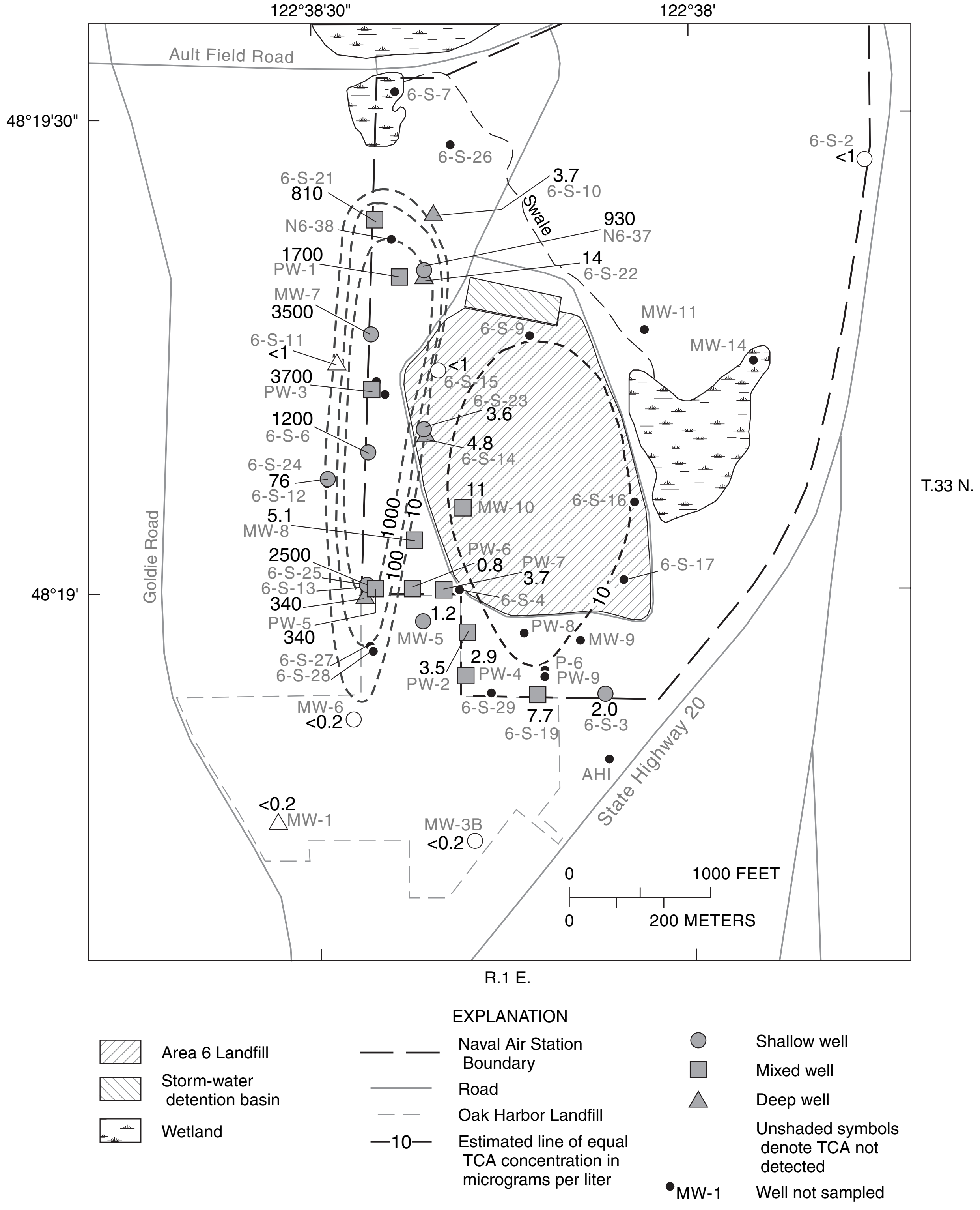

Figure 10b. Concentrations of 1,1,1-trichloroethane (TCA) in shallow ground water at Area 6, 1995. Units are micrograms per liter.

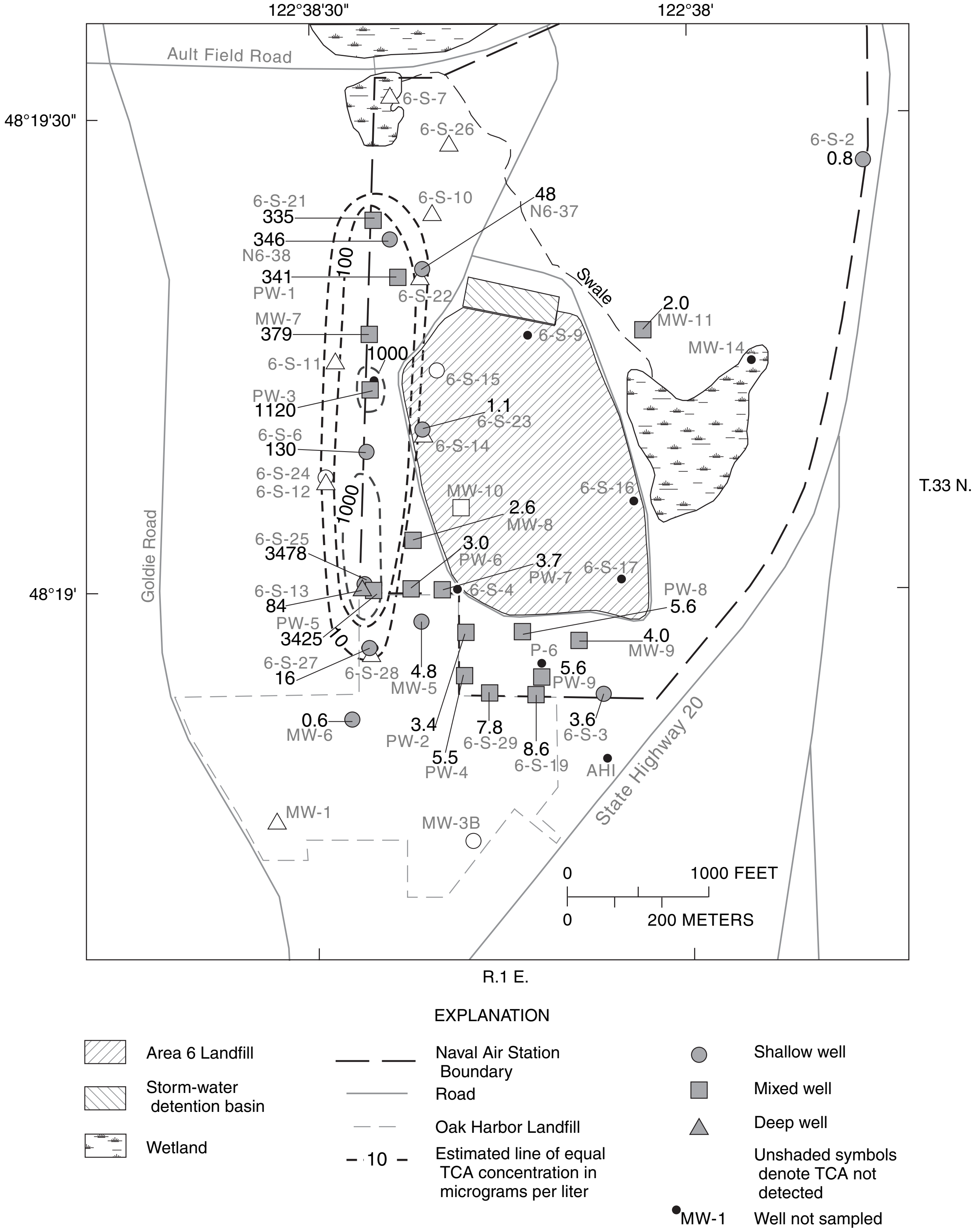

Figure 10c. Concentrations of 1,1,1-trichloroethane (TCA) in shallow ground water at Area 6, 1998. Units are micrograms per liter. Unshaded symbols denote TCA not detected at 0.5 microgram per liter.

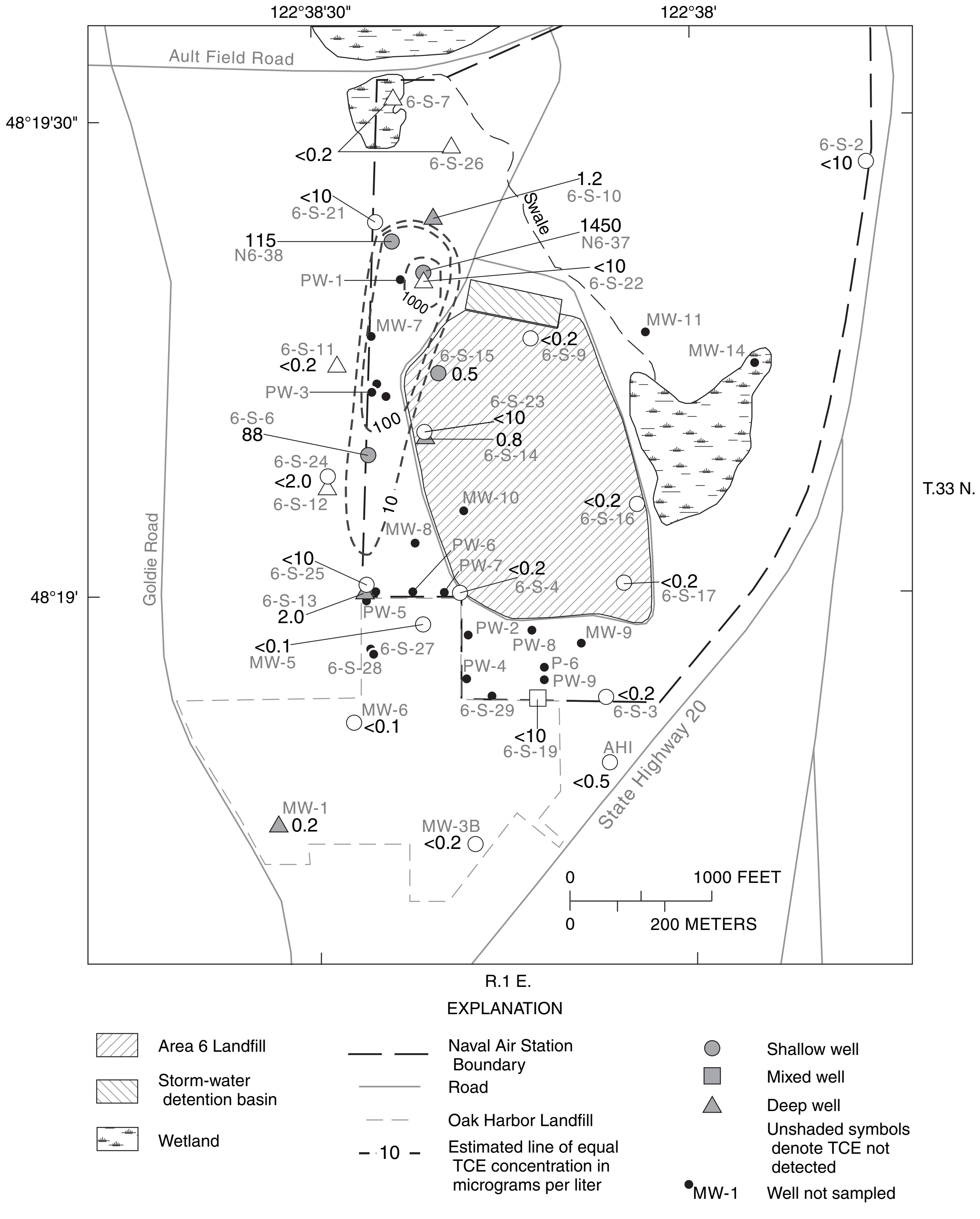

Figure 11a. Concentrations of trichloroethene (TCE) in shallow ground water at Area 6, 1991. Units are micrograms per liter.

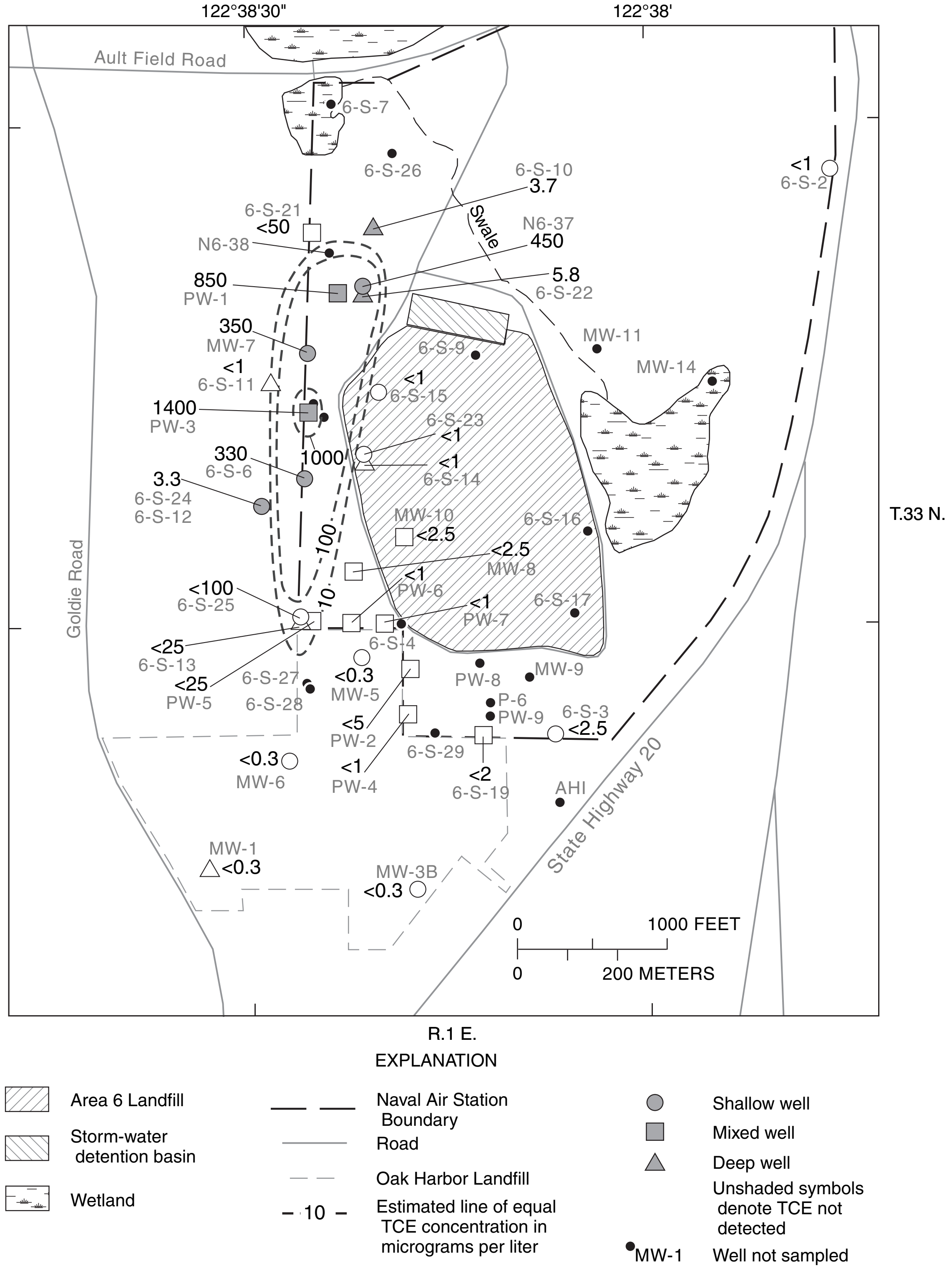

Figure 11b. Concentrations of trichloroethene (TCE) in shallow ground water at Area 6, 1995. Units are micrograms per liter.

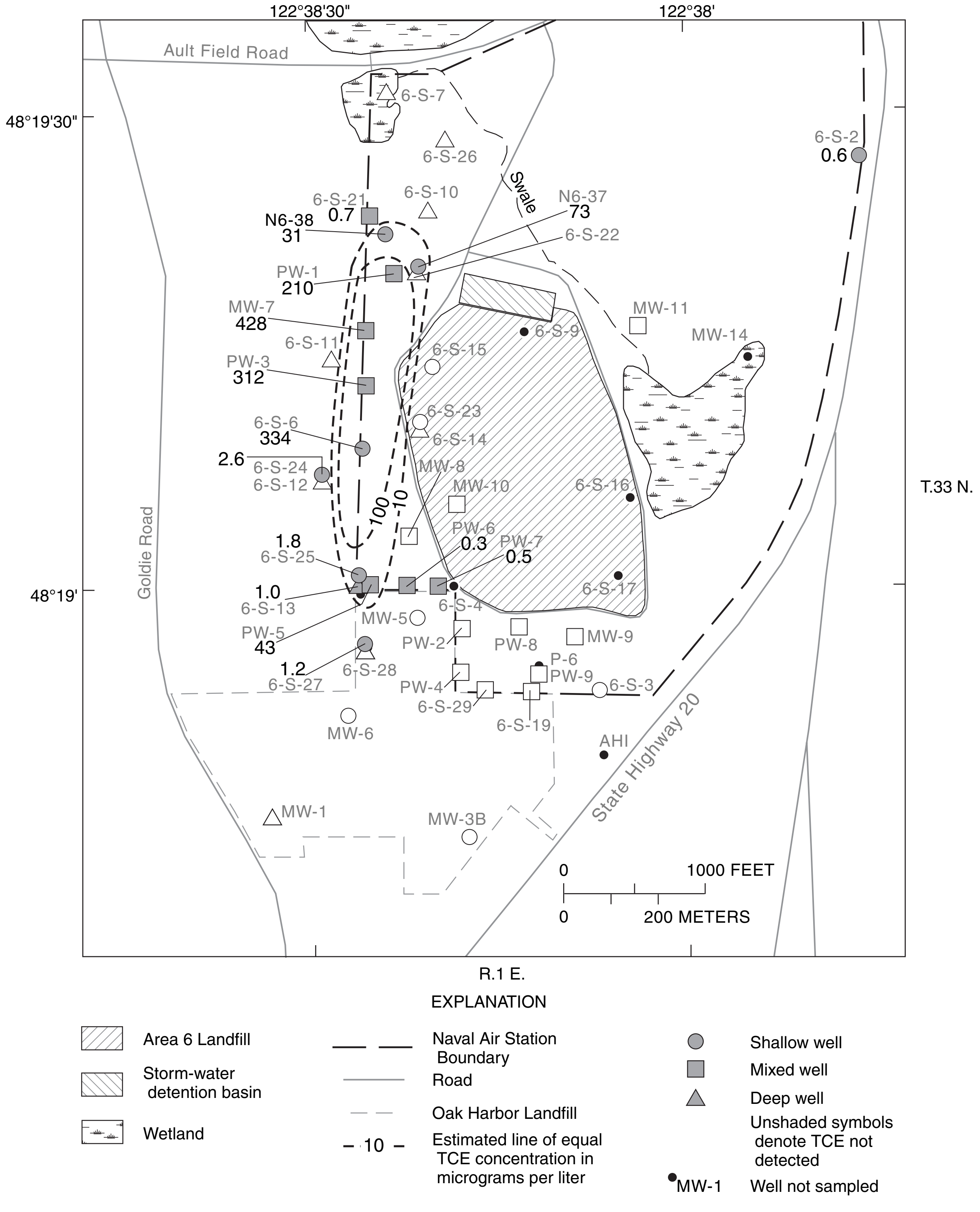

Figure 11c. Concentrations of trichloroethene (TCE) in shallow ground water at Area 6, 1998. Units are micrograms per liter. Unshaded symbols denote TCE not detected at 0.2 microgram per liter.

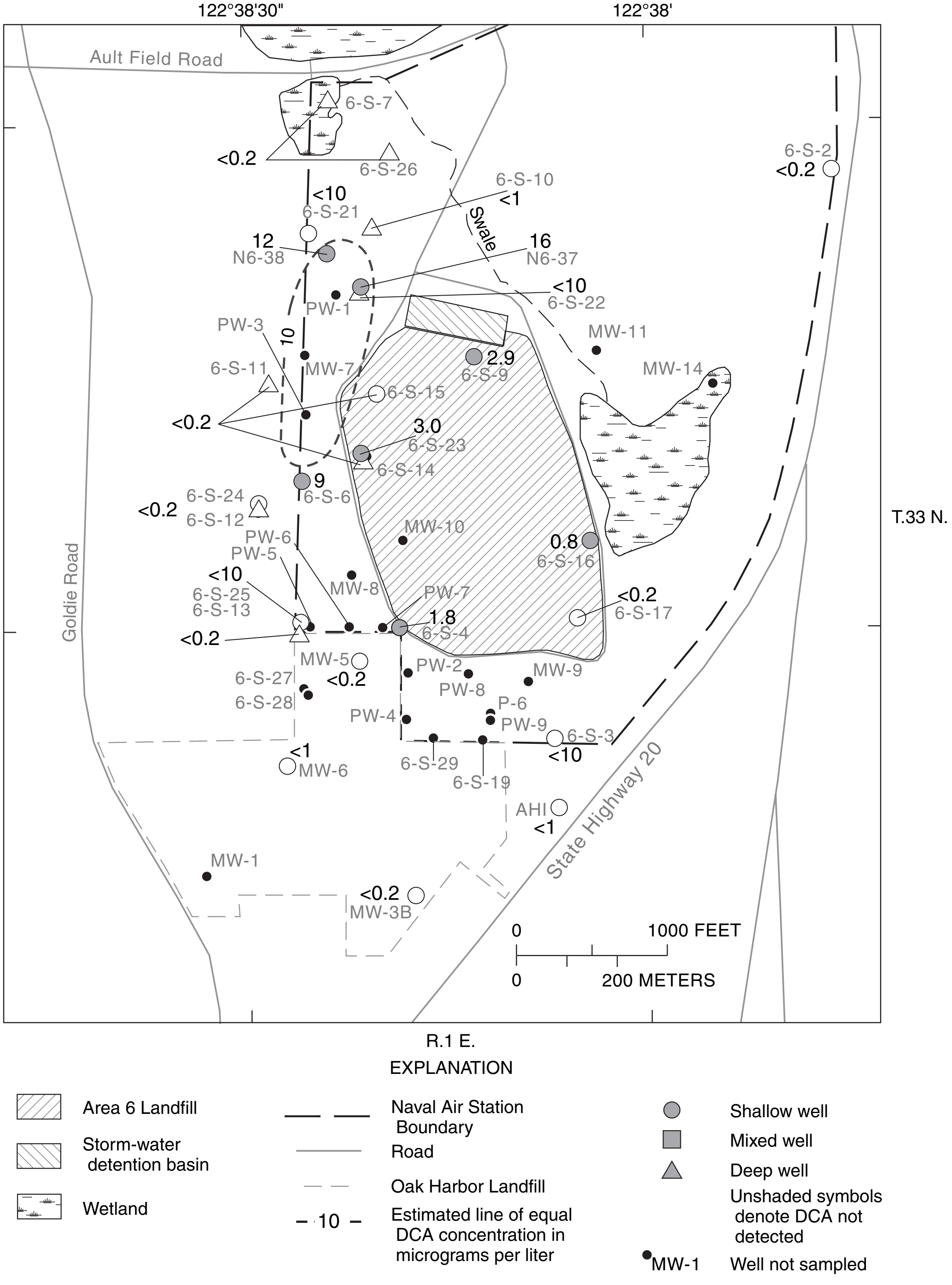

Figure 12a. Concentrations of 1,1-dichloroethane (DCA) in shallow ground water at Area 6, 1991. Units are micrograms per liter.

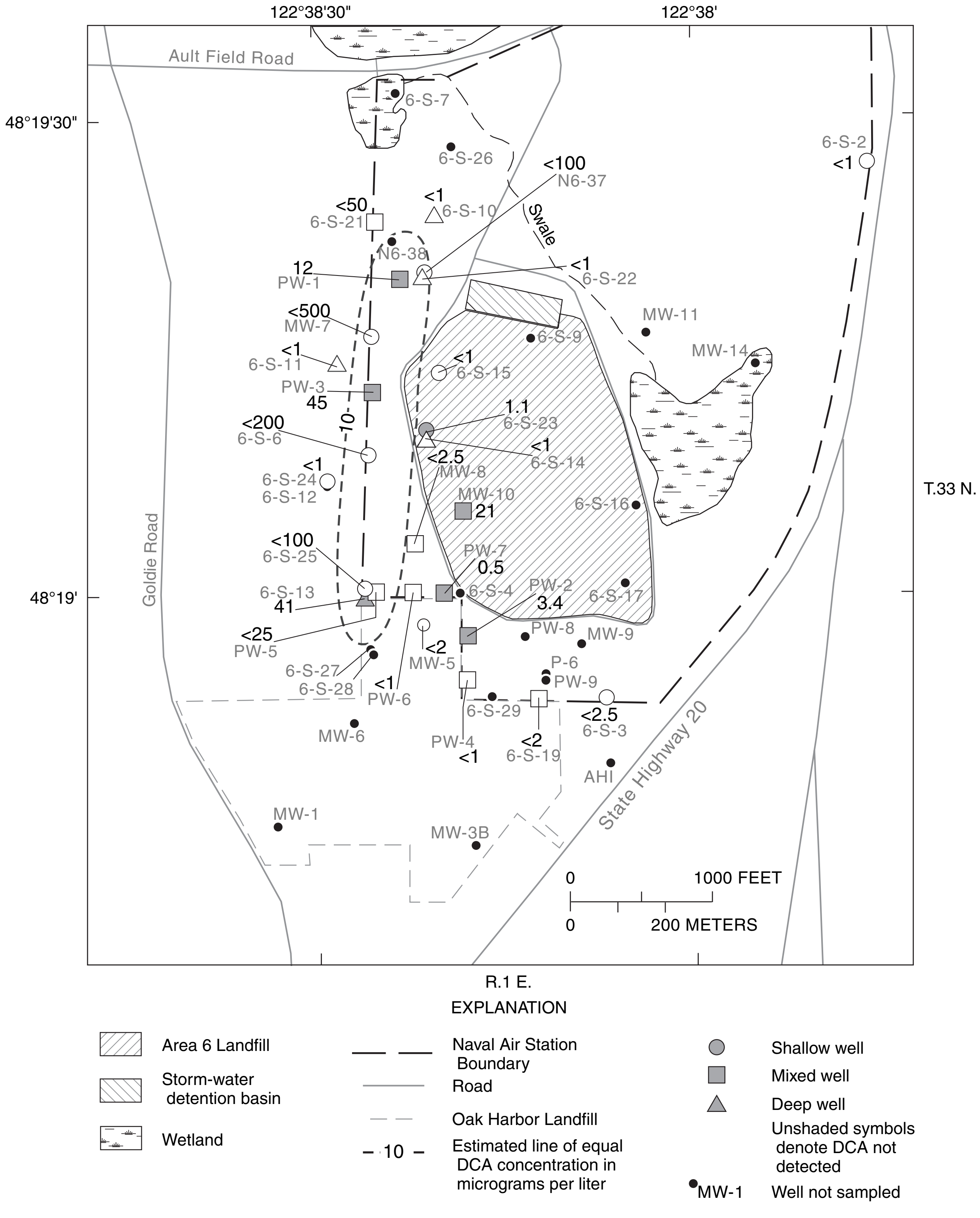

Figure 12b. Concentrations of 1,1-dichloroethane (DCA) in shallow ground water at Area 6, 1995. Units are micrograms per liter.

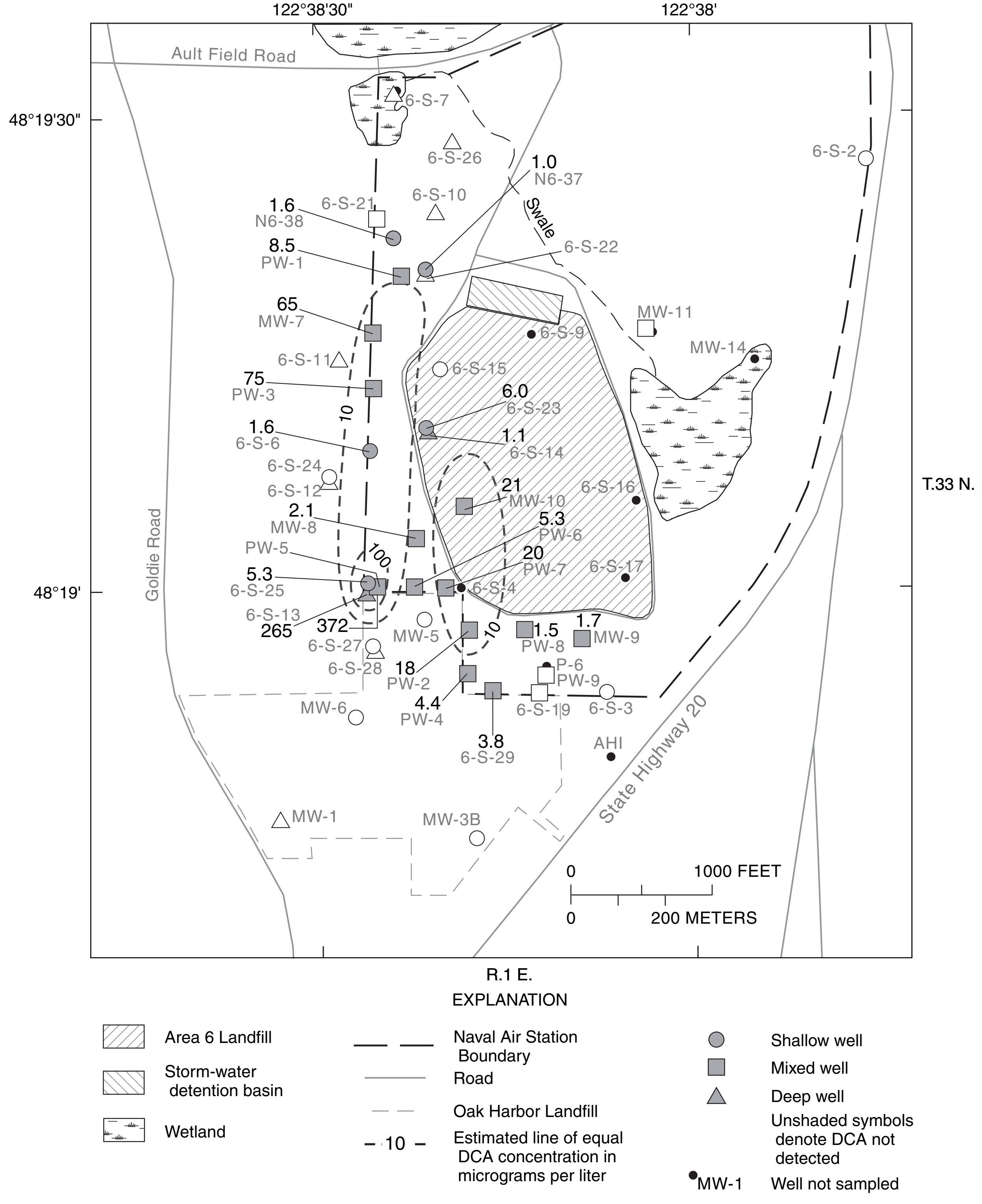

Figure 12c. Concentrations of 1,1-dichloroethane (DCA) in shallow ground water at Area 6, 1998. Units are micrograms per liter. Unshaded symbols denote DCA not detected at 0.2 microgram per liter.

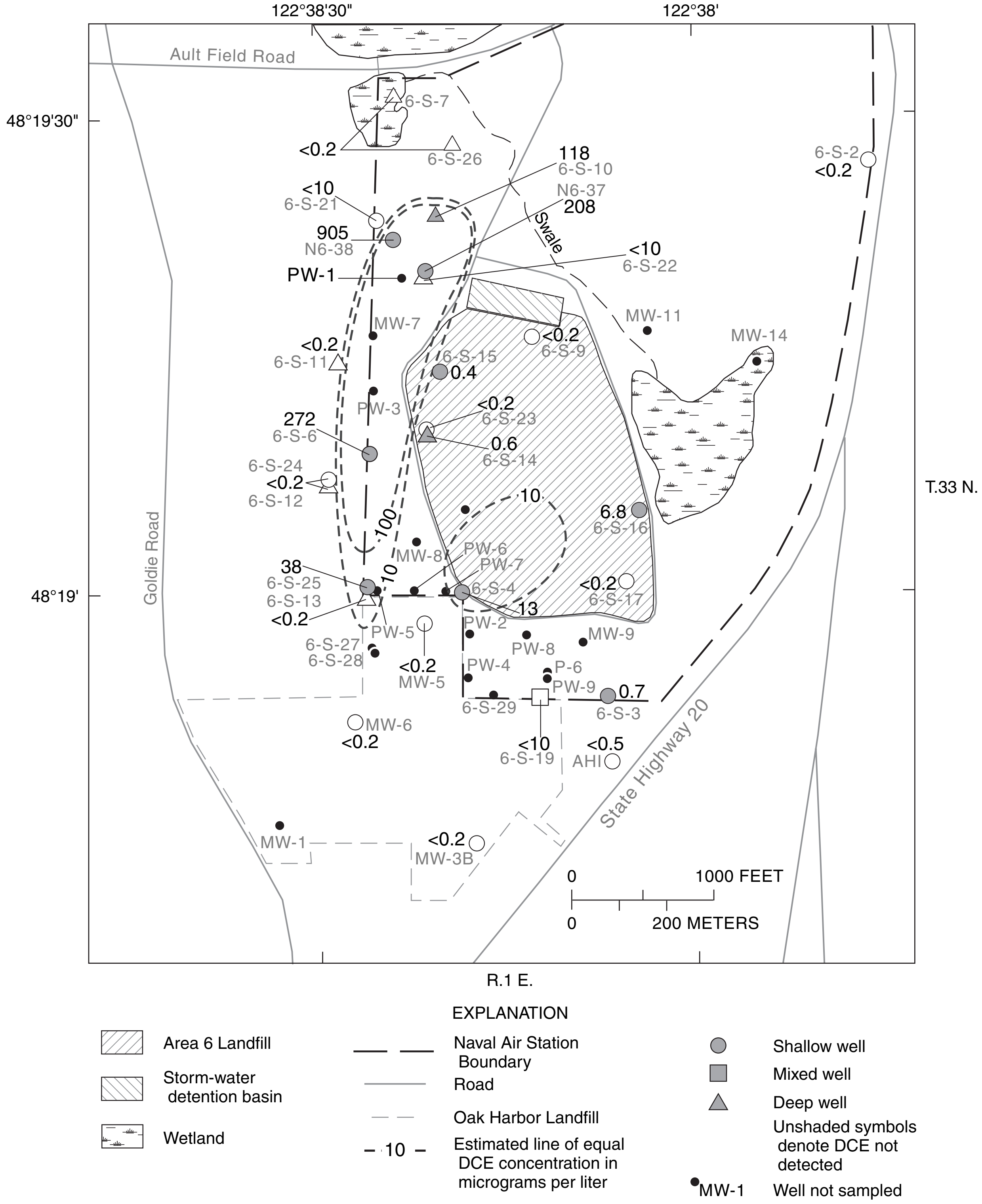

Figure 13a. Concentrations of 1,1-dichloroethene (DCE) in shallow ground water at Area 6, 1991. Units are micrograms per liter.

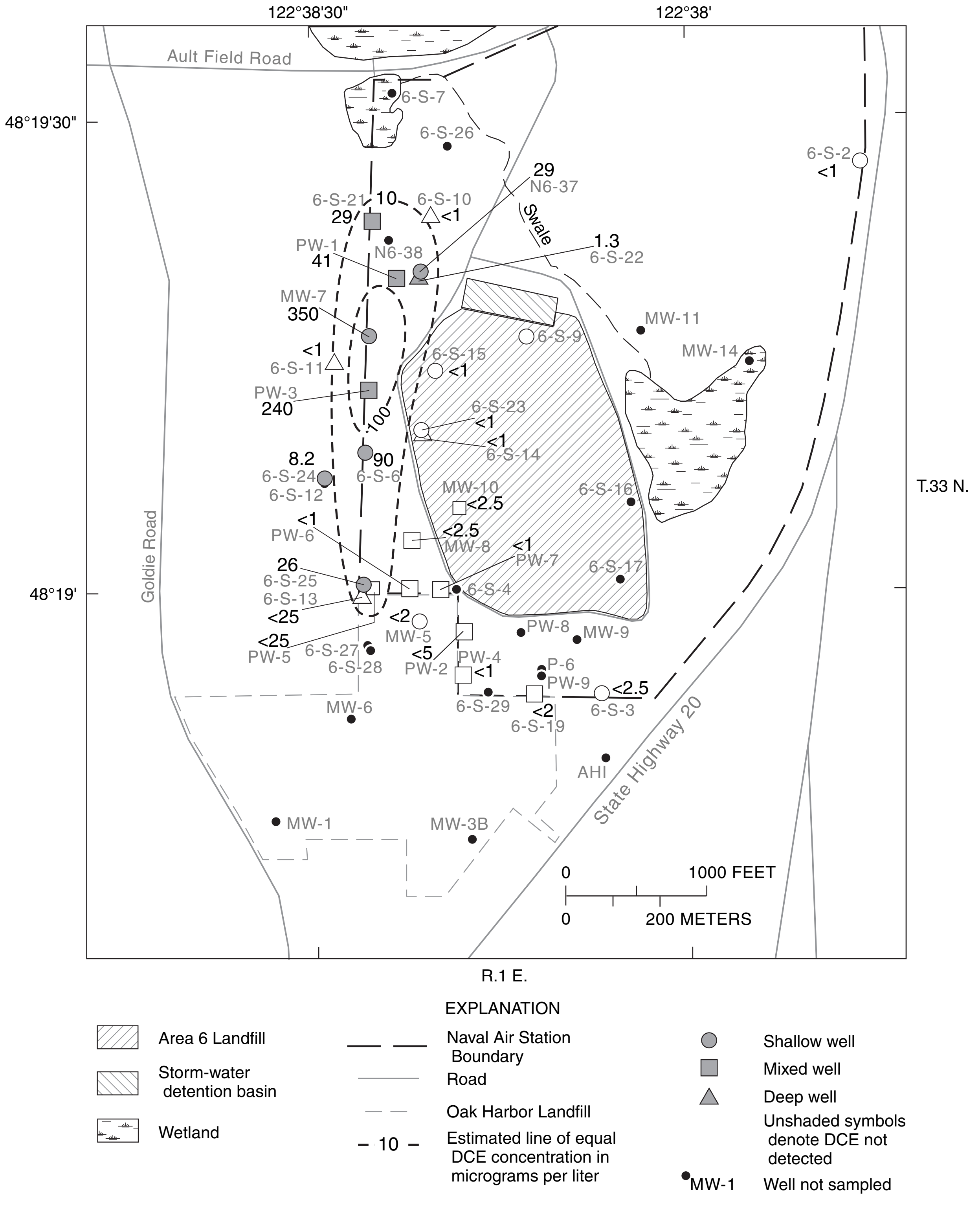

Figure 13b. Concentration of 1,1-dichloroethene (DCE) in shallow ground water at Area 6, 1995. Units are micrograms per liter.

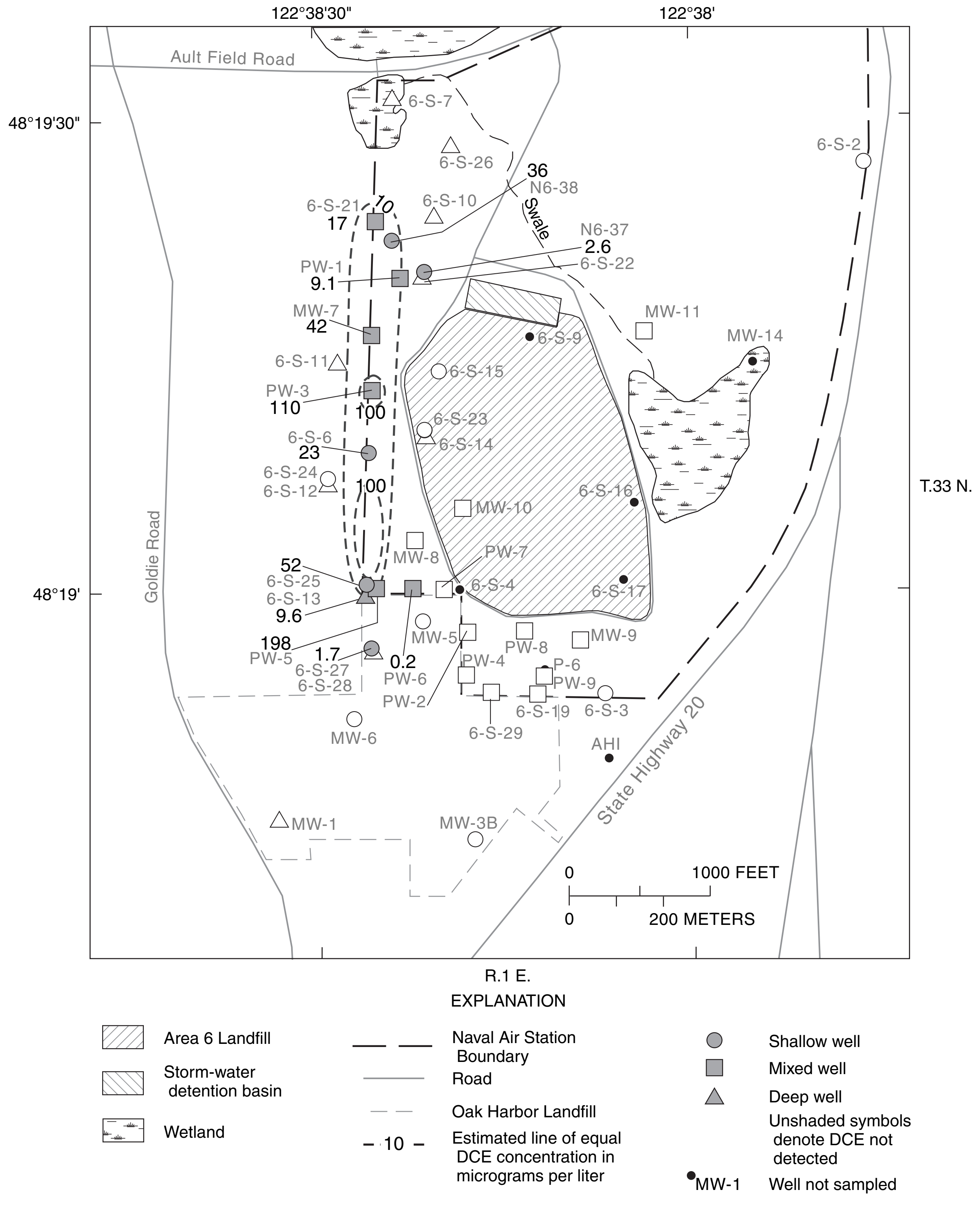

Figure 13c. Concentrations of 1,1-dichloroethene (DCE) in shallow ground water at Area 6, 1998. Units are micrograms per liter. Unshaded symbols denote DCE not detected at 0.2 microgram per liter.

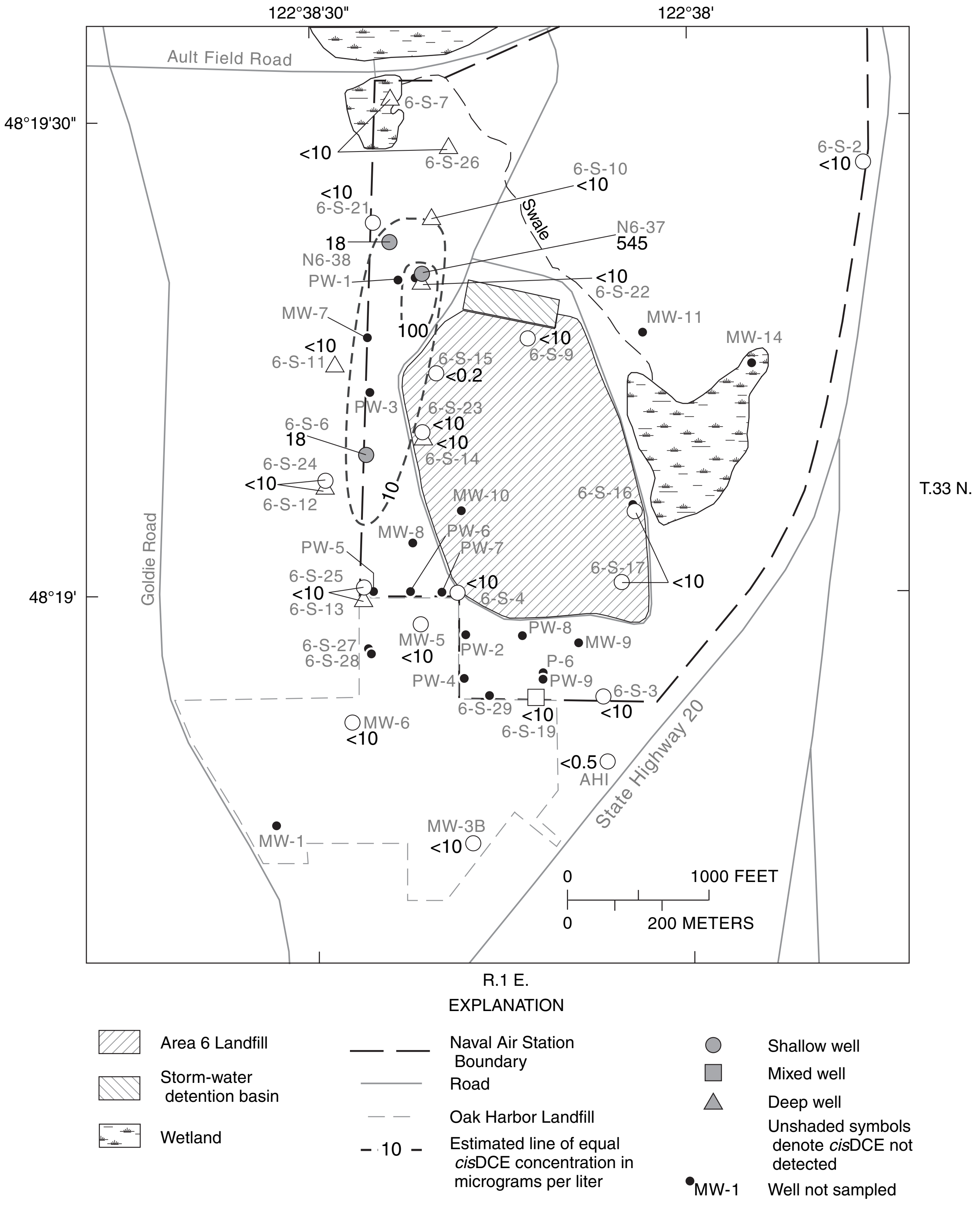

Figure 14a. Concentrations of *cis*-1,2-dichloroethene (*cis*DCE) in shallow ground water at Area 6, 1991. Units are micrograms per liter.

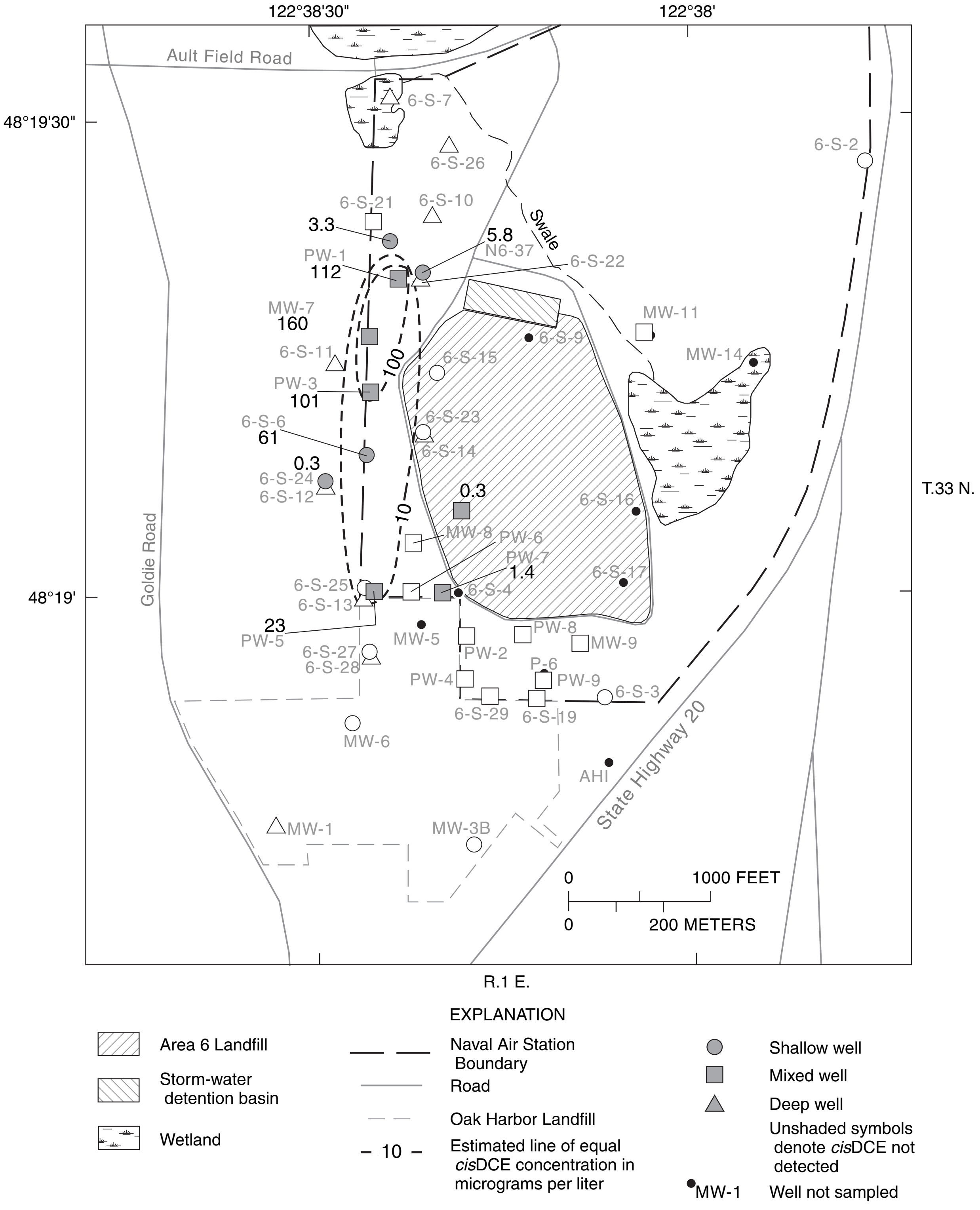

Figure 14b. Concentrations of *cis*-1,2-dichloroethene (*cis*DCE) in shallow ground water at Area 6, 1998. Units are micrograms per liter. Unshaded symbols denote *cis*DCE not detected at 0.2 microgram per liter.

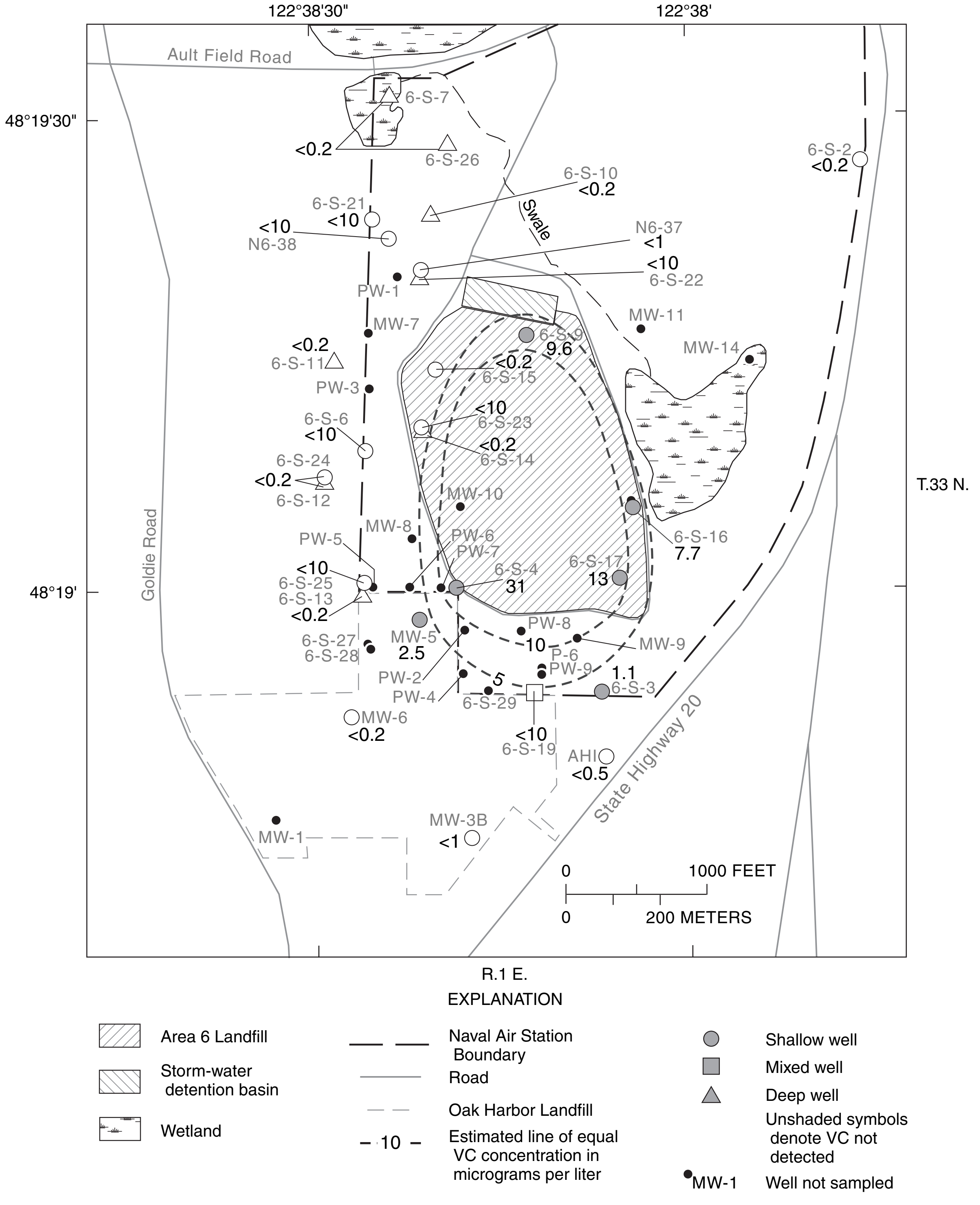

Figure 15a. Concentrations of vinyl chloride (VC) in shallow ground water at Area 6, 1991. Units are micrograms per liter.

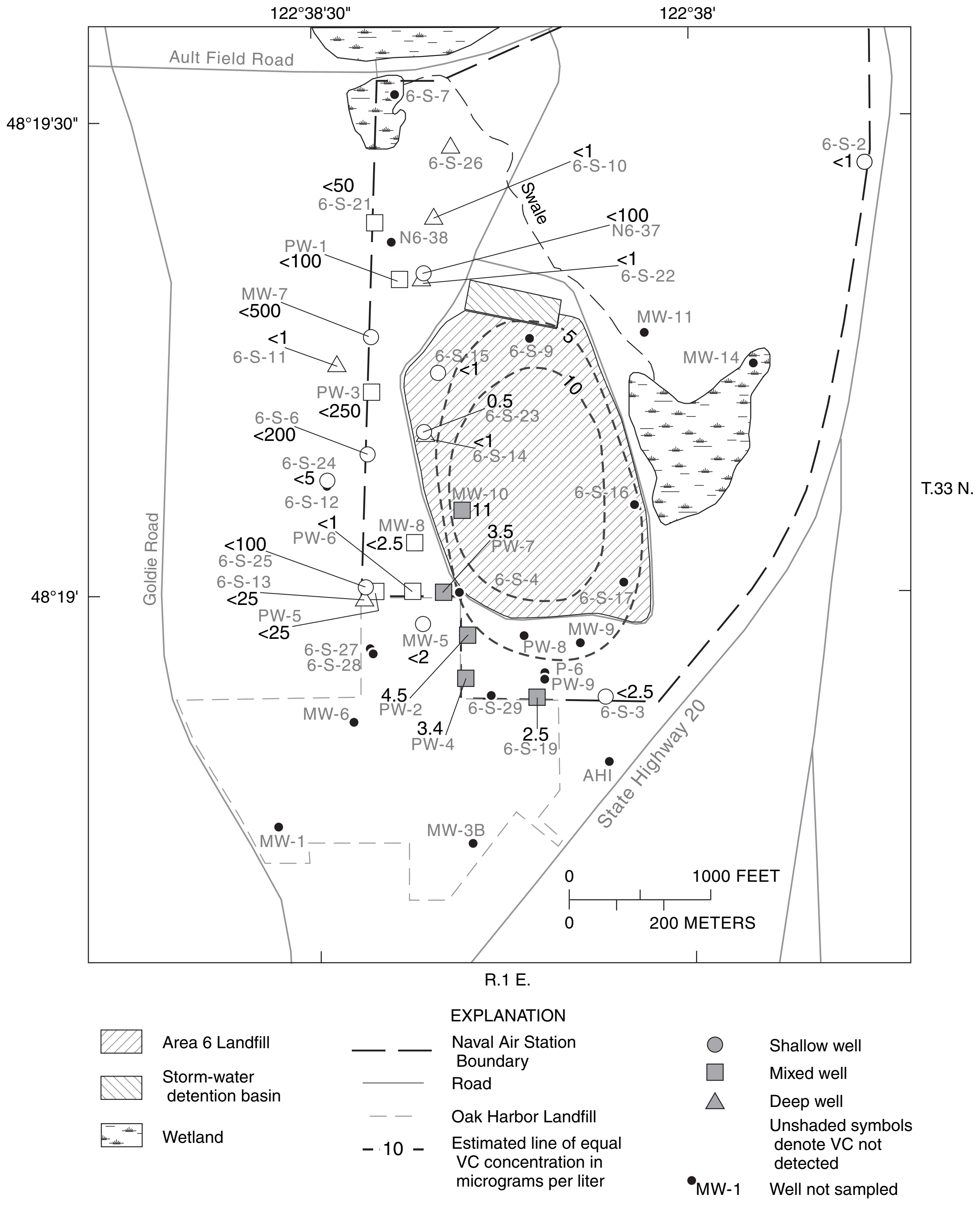

Figure 15b. Concentrations of vinyl chloride (VC) in shallow ground water at Area 6, 1995. Units are micrograms per liter.

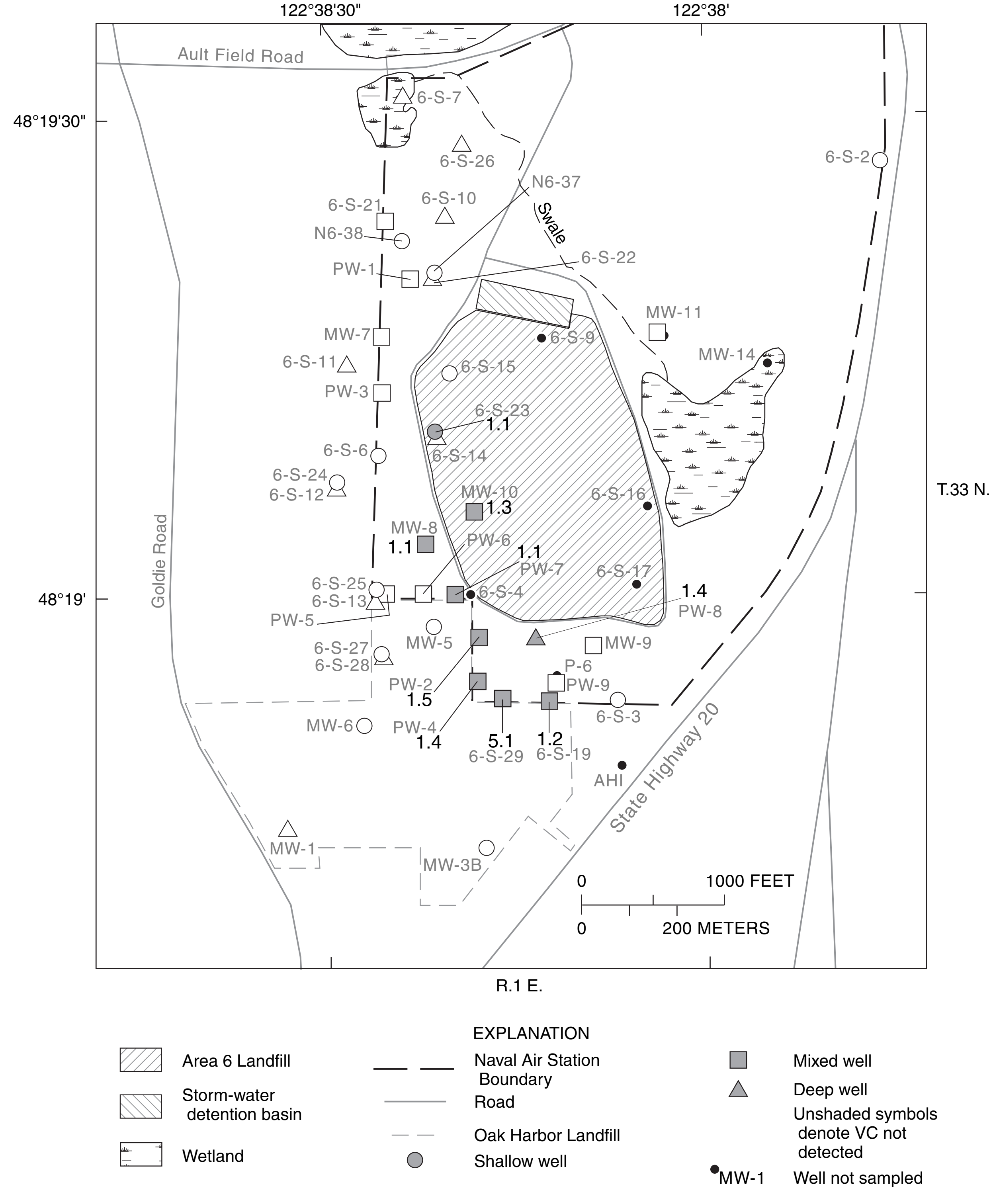

Figure 15c. Concentrations of vinyl chloride (VC) in shallow ground water at Area 6, 1998. Units are micrograms per liter. Unshaded symbols denote VC not detected.

Table 4. Cumulative summary of chlorinated volatile organic compounds (VOC) data collected at Area 6 Naval Air Station Whidbey Island extraction wells and monitoring wells (from Foster Wheeler Environmental, 1999)

[Results are reported in micrograms per liter (µg/L); Effluent limitations are shown on this table for comparison to ground-water quality criteria; 1995 and later samples analyzed by U.S. Environmental Protection Agency Method 601; 1993 samples analyzed by U.S. Environmental Protection Agency Method 5242; duplicate samples for a given well are labeled with the same date; **TCE**, trichloroethene; **TCA**, 1,1,1-trichloroethane; **DCA**, 1,1-dichloroethane; ***cis*-DCE**, *cis*-1,2-dichloroethene; **DCE**, 1,1,-dicloroethene; **VC**, vinyl chloride; ND ( ), indicates parameter not detected with detection limit in parenthesis; NR, no reading; –, not analyzed for indicated parameter; J, estimated value—detected, but below quantitation limit; **bold** indicates detected values]

Well identification number	Sample date	Parameters						
		pH	TCE	TCA	DCA	*cis*-DCE	DCE	VC
Effluent limitation		6.5 to 8.5	5	200	800	70	0.07	0.1
PW-1	08-25-93	7.20	**1,400**	**2,800**	ND (200)	–	ND (200)	ND (200)
	02-10-95	6.96	**850**	**1,700**	**12 J**	–	**41 J**	ND (100)
	02-10-95	6.96	**830**	**1,600**	ND (120)	–	**41 J**	ND (120)
	09-19-95	7.48	**480**	**1,700**	ND (50)	–	**62**	ND (50)
	12-04-95	8.00	**350**	**1,000**	ND (50)	–	**49 J**	ND (50)
	03-25-96	8.68	**270**	**890**	ND (50)	–	**30 J**	ND (50)
	11-01-96	5.81	**290**	**420**	**4.7 J**	–	**12 J**	ND (25)
	01-28-97	6.99	**390**	**320**	ND (25)	–	ND (25)	ND (25)
	05-01-97	7.67	**320**	**230**	ND (10)	**120**	**8.1 J**	ND (10)
	05-01-97	7.67	**310**	**240**	**7.6 J**	**130**	ND (10)	ND (10)
	07-21-97	7.20	**290**	**220**	ND (10)	**79**	**4.5 J**	ND (10)
	10-28-97	6.52	**300**	**290**	**7.1**	**130**	**11**	ND (0.20)
	10-28-97	6.52	**300**	**280**	**7.5**	**140**	**11**	ND (0.20)
	01-20-98	6.94	**220**	**320**	**11**	**150**	**13**	ND (1.0)
	01-20-98	6.94	**200**	**310**	**10**	**140**	**12**	ND (1.0)
	04-16-98	6.7	**150**	**320**	**10**	**110**	**9.5**	ND (1.0)
	07-29-98	6.5	**250**	**380**	**7.8**	**110**	**12**	ND (1.0)
	07-29-98	6.5	**240**	**380**	**7.5**	**110**	**12**	ND (1.0)
	10-27-98	6.7	**240**	**400**	**6.0**	**88**	**13**	ND (1.0)
	10-27-98	6.7	**230**	**300**	**5.4**	**83**	**12**	ND (1.0)
PW-2	08-30-93	7.50	ND (1)	**3.6**	**2.3**	–	ND (1)	**4.8**
	06-23-95	6.84	ND (5)	**3.5 J**	**3.4 J**	–	ND (5)	**4.5 J**
	09-19-95	6.83	ND (5)	ND (5)	**2.2 J**	–	ND (5)	**5.9**
	12-05-95	7.43	ND (5)	**5.4**	**7.6**	–	ND (5)	**12**
	03-25-96	6.9	ND (2)	**7.3**	**7.2**	–	ND (2)	**7.8**
	11-01-96	7.48	ND (5.0)	**4.4 J**	**11**	–	ND (5.0)	ND (5.0)
	01-29-97	6.91	ND (2.0)	**5.1**	**13**	–	ND (2.0)	**0.98 J**
	05-01-97	6.4	ND (2.0)	**4.8**	**13**	ND (2.0)	ND (2.0)	**1.8 J**
	07-21-97	7.1	ND (2.0)	**5.6**	**12**	ND (2.0)	ND (2.0)	**3.4**
	04-16-98	6.8	ND (2.0)	**3.4**	**18**	ND (2.0)	ND (2.0)	**1.5**
PW-3	08-29-93	7.50	**1,900**	**4,600**	ND (500)	–	ND (500)	ND (500)
	02-10-95	7.12	**1,400**	**3,700**	**45 J**	–	**240 J**	ND (250)
	09-19-95	7.25	**870**	**4,500**	ND (200)	–	**410**	ND (200)
	09-19-95	7.72	**770**	**4,500**	**37 J**	–	**400**	ND (200)
	12-04-95	7.88	**700**	**3,400**	**150 J**	–	**340**	ND (200)
	03-25-96	7.15	**550**	**3,200**	**110**	–	**300**	ND (200)
	03-25-96	7.13	**500**	**2,900**	**110**	–	**330**	ND (100)
	11-01-96	7.74	**560**	**2,300**	**79 J**	–	**180**	ND (100)
	11-01-96	7.74	**510**	**2,200**	**110**	–	**170**	ND (100)
	01-28-97	7.31	**620**	**1,300**	**57 J**	–	**77 J**	ND (100)

Table 4. Cumulative summary of chlorinated volatile organic compounds (VOC) data collected at Area 6 Naval Air Station Whidbey Island extraction wells and monitoring wells (from Foster Wheeler Environmental, 1999)—*Continued*

[Results are reported in micrograms per liter (µg/L); Effluent limitations are shown on this table for comparison to ground-water quality criteria; 1995 and later samples analyzed by U.S. Environmental Protection Agency Method 601; 1993 samples analyzed by U.S. Environmental Protection Agency Method 5242; duplicate samples for a given well are labeled with the same date; **TCE**, trichloroethene; **TCA**, 1,1,1-trichloroethane; **DCA**, 1,1-dichloroethane; ***cis*-DCE**, *cis*-1,2-dichloroethene; **DCE**, 1,1,-dicloroethene; **VC**, vinyl chloride; ND ( ), indicates parameter not detected with detection limit in parenthesis; NR, no reading; –, not analyzed for indicated parameter; J, estimated value—detected, but below quantitation limit; **bold** indicates detected values]

Well identification number	Sample date	Parameters						
		pH	TCE	TCA	DCA	*cis*-DCE	DCE	VC
PW-3	05-01-97	6.58	**420**	**930**	**73**	**120**	**150**	ND (25)
	07-21-97	7.3	**440**	**1,000**	**65**	**120**	**120**	ND (25)
	10-14-97	6.72	**440**	**590**	**67**	**140**	**83**	**0.30**
	10-27-97	–	–	–	–	–	–	–
	01-20-98	7.06	**280**	**1,300**	**79**	**109**	**120**	ND (1)
	04-16-98	6.8	**200**	**780**	**71**	**92 J**	**78**	ND (1.0)
	07-29-98	6.6	**400**	**1,100**	**81**	**110**	**110**	ND (5.0)
	10-27-98	6.8	**370**	**1,300**	**70**	**92**	**130**	ND (1.0)
PW-4	08-27-93	7.40	ND (1)	ND (1)	ND (1)	–	ND (1)	**2.9**
	02-10-95	6.95	ND (1)	**2.9**	ND (1)	–	ND (1)	**3.4**
	09-19-95	6.70	ND (5)	ND (5)	ND (5)	–	ND (5)	**3.2 J**
	12-04-95	7.91	ND (5)	**3.0 J**	**1.9 J**	–	ND (5)	ND (5)
	03-25-96	7.23	ND (2)	**3.0**	**0.48 J**	–	ND (2)	**4.7**
	11-01-96	7.55	ND (5.0)	**3.1 J**	**0.93 J**	–	ND (5.0)	**1.7 J**
	01-29-97	6.87	ND (2.0)	**4.2 J**	**0.95 J**	–	ND (2.0)	ND (2.0)
	05-01-97	6.47	ND (2.0)	**3**	**1.2 J**	ND (2.0)	ND (2.0)	**2.6**
	07-21-97	7.1	ND (2.0)	**2.6**	**1.4 J**	ND (2.0)	ND (2.0)	**1.6 J**
	10-27-97	7.29	ND (0.2)	**5.3**	**3.1**	ND (0.20)	ND (0.20)	**1.6**
	01-20-98	7.17	ND (0.2)	**5.1**	**4.1**	ND (0.20)	ND (0.20)	**1.6**
	04-16-98	6.9	ND (0.2)	**9.2**	**4.7**	ND (0.20)	ND (0.20)	**1.7**
	07-29-98	6.6	ND (0.2)	**4.1**	**4.5**	ND (0.20)	ND (0.20)	ND (1.0)
	10-27-98	6.8	ND (2.0)	**3.7**	**4.1**	ND (0.20)	ND (0.20)	**1.1**
PW-5	08-29-93	7.89	ND (1)	**130**	ND (1)	–	ND (1)	ND (1)
	02-10-95	7.35	ND (25)	**340**	ND (25)	–	ND (25)	ND (25)
	09-19-95	7.48	ND (100)	**2,100**	**57 J**	–	ND (100)	ND (100)
	12-04-95	8.31	ND (100)	**2,400**	**240**	–	**72 J**	ND (100)
	12-04-95	8.31	ND (100)	**2,400**	**230**	–	**74 J**	ND (100)
	03-25-96	7.98	ND (100)	**2,700**	**330**	–	**42 J**	ND (100)
	11-01-96	7.99	ND (250)	**2,800**	**280**	–	**55 J**	ND (250)
	01-28-97	7.65	ND (250)	**3,300**	**290**	–	ND (250)	ND (250)
	05-01-97	7.20	ND (100)	**2,800**	**330**	ND (100)	**130**	ND (100)
	07-21-97	7.70	ND (100)	**3,000**	**280**	ND (100)	**120**	ND (100)
	10-14-97	7.30	**10**	**3,100**	**230**	**14**	**130**	**0.47**
	10-27-97	–	–	–	–	–	–	–
	01-20-98	7.72	**29**	**4,300**	**370**	**21**	**210**	ND (1.0)
	04-16-98	7.2	**29**	**2,600**	**390**	**19**	**130**	ND (1.0)
	07-29-98	7.0	**37**	**3,300**	**320**	**24**	**190**	ND (1.0)
	10-27-98	7.2	**76**	**3,500**	**410**	**27**	**260**	ND (1.0)
PW-6	08-30-93	8.01	ND (1)	**1.1**	ND (1)	–	ND (1)	ND (1)
	02-10-95	6.97	ND (1)	**0.78 J**	ND (1)	–	ND (1)	ND (1)
	09-19-95	7.15	ND (2)	ND (2)	ND (2)	–	ND (2)	ND (2)
	12-04-95	7.95	ND (5)	**1.9 J**	ND (5)	–	ND (5)	ND (5)
	03-25-96	7.36	ND (1)	**1.3**	ND (1)	–	ND (1)	ND (1)

Table 4. Cumulative summary of chlorinated volatile organic compounds (VOC) data collected at Area 6 Naval Air Station Whidbey Island extraction wells and monitoring wells (from Foster Wheeler Environmental, 1999)—*Continued*

[Results are reported in micrograms per liter (µg/L); Effluent limitations are shown on this table for comparison to ground-water quality criteria; 1995 and later samples analyzed by U.S. Environmental Protection Agency Method 601; 1993 samples analyzed by U.S. Environmental Protection Agency Method 5242; duplicate samples for a given well are labeled with the same date; **TCE**, trichloroethene; **TCA**, 1,1,1-trichloroethane; **DCA**, 1,1-dichloroethane; ***cis*-DCE**, *cis*-1,2-dichloroethene; **DCE**, 1,1,-dicloroethene; **VC**, vinyl chloride; ND ( ), indicates parameter not detected with detection limit in parenthesis; NR, no reading; –, not analyzed for indicated parameter; J, estimated value—detected, but below quantitation limit; **bold** indicates detected values]

Well identification number	Sample date	Parameters						
		pH	TCE	TCA	DCA	*cis*-DCE	DCE	VC
	11-01-96	7.81	ND (2.0)	**1.4 J**	ND (2.0)	–	ND (2.0)	ND (2.0)
	01-28-97	7.35	ND (2.0)	**1.5 J**	ND (2.0)	–	ND (2.0)	ND (2.0)
	05-01-97	6.70	ND (1.0)	**0.52 J**	ND (1.0)	ND (1.0)	ND (1.0)	**0.30 J**
	07-21-97	7.20	ND (1.0)	**1.3**	ND (1.0)	ND (1.0)	ND (1.0)	**0.43 J**
	10-27-97	7.17	ND (0.20)	**2.0**	ND (0.20)	ND (0.20)	ND (0.20)	**0.50**
	01-20-98	7.18	**0.70**	**7.3**	**0.32**	ND (0.20)	**0.37**	ND (1)
	04-16-98	6.8	ND (0.20)	ND (0.20)	**20**	ND (0.20)	ND (0.20)	ND (0.20)
	07-29-98	6.6	ND (0.20)	**2.4**	**0.38**	ND (0.20)	ND (0.20)	ND (1.0)
	10-27-98	6.8	ND (0.20)	**2.0**	**0.38**	ND (0.20)	ND (0.20)	ND (1.0)
PW-7	08-27-93	7.28	ND (1)	**3.0**	ND (1)	–	ND (1)	**4**
	02-10-95	6.90	ND (1)	**3.7**	**0.52 J**	–	ND (1)	**3.5**
	09-19-95	6.66	ND (5)	ND (5)	**1.9 J**	–	ND (5)	**4.2 J**
	12-04-95	7.70	ND (5)	**3.8 J**	**5.4**	–	ND (5)	ND (5)
	03-25-96	7.47	ND (2)	**3.8**	**8.0**	–	ND (2)	**8.8**
	11-01-96	NR	ND (5.0)	**2.3 J**	**10**	–	ND (5.0)	**1.7 J**
	05-01-97	6.48	ND (5.0)	**1.1 J**	**15**	ND (5.0)	ND (5.0)	**2.0 J**
	07-21-97	7.20	ND (5.0)	**1.2 J**	**12**	ND (5.0)	ND (5.0)	**1.9 J**
	10-27-97	7.19	ND (0.20)	**2.8**	**18**	**0.97**	ND (0.20)	**1.5**
	01-20-98	7.05	**1.6**	**8.8**	**20**	**1.9**	ND (0.20)	**1.0**
	04-16-98	6.7	**0.21**	**2.5**	**23**	**1.7**	ND (0.20)	**1.5**
	04-16-98	6.7	ND (0.20)	ND (0.50)	ND (0.20)	ND (0.20)	ND (0.20)	ND (1.0)
	07-29-98	6.6	ND (0.20)	**1.7**	**20**	**1.0**	ND (0.20)	ND (1.0)
	10-27-98	6.8	ND (0.20)	**1.8**	**19**	**1.2**	ND (0.20)	ND (1.0)
PW-8	09-18-96	7.02	ND (2.0)	**8.1**	ND (2.0)	–	ND (2.0)	**2.3**
	11-01-96	7.46	ND (5.0)	**3.9 J**	ND (5.0)	–	ND (5.0)	ND (5.0)
	01-29-97	6.80	ND (1.0)	**2.4**	ND (1.0)	–	ND (1.0)	**0.88 J**
	05-03-97	6.43	ND (2.0)	**5.2**	ND (2.0)	ND (2.0)	ND (2.0)	**2.7**
	07-21-97	6.70	ND (2.0)	**5.0**	ND (2.0)	ND (2.0)	ND (2.0)	**2.7**
	10-27-97	7.08	**0.26**	**6.9**	**0.60**	ND (0.20)	ND (0.20)	**2.0**
	01-20-98	7.02	ND (0.20)	**7.3**	**0.93**	ND (0.20)	**0.21**	**1.9**
	04-16-98	6.8	ND (0.20)	ND (0.50)	ND (0.20)	ND (0.20)	ND (0.20)	ND (1.0)
	07-29-98	6.6	ND (0.20)	**9.6**	**2.0**	ND (0.20)	ND (0.20)	**1.8**
	10-27-98	6.5	ND (0.20)	**4.9**	**2.8**	ND (0.20)	ND (0.20)	ND (1.0)
PW9	09-19-96	7.37	ND (2.0)	**3.7**	ND (2.0)	–	ND (2.0)	ND (2.0)
	11-01-96	7.61	ND (2.5)	**3.7**	ND (2.5)	–	ND (2.5)	ND (2.5)
	01-29-97	6.97	ND (2.5)	**2.6**	ND (2.5)	–	ND (2.5)	ND (2.5)
	05-01-97	6.62	ND (2.5)	**4.3**	ND (2.5)	ND (2.5)	ND (2.5)	**1.0 J**
	07-21-97	6.70	ND (2.0)	**4.3**	ND (2.5)	ND (2.5)	ND (2.5)	**1.2 J**
	07-21-97	6.70	ND (2.5)	**3.9**	ND (2.5)	ND (2.5)	ND (2.5)	**1.4 J**
	10-27-97	7.14	ND (0.20)	**6.3**	ND (0.20)	ND (0.20)	ND (0.20)	**0.81**
	01-20-98	7.06	ND (0.20)	**6.6**	ND (0.20)	ND (0.20)	**0.20**	ND (1.0)
	07-29-98	6.6	ND (0.20)	**5.6**	ND (0.20)	ND (0.20)	ND (0.20)	ND (1.0)
	10-27-98	6.8	ND (0.20)	**4.5**	ND (0.20)	ND (0.20)	ND (0.20)	ND (1.0)

Table 4. Cumulative summary of chlorinated volatile organic compounds (VOC) data collected at Area 6 Naval Air Station Whidbey Island extraction wells and monitoring wells (from Foster Wheeler Environmental, 1999)—*Continued*

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Well identification number	Sample date	Parameters						
		pH	TCE	TCA	DCA	*cis*-DCE	DCE	VC
6-S-2	02-11-95	7.7	ND (1) J	ND (1)	ND (1)	–	ND (1)	ND (1)
	09-24-95	7.9	ND (1) J	**0.50 J**	ND (1)	–	ND (1)	ND (1)
	12-06-95	7.8	**0.28 J**	**1.3**	ND (1)	–	ND (1)	ND (1)
	03-26-96	7.6	ND (1)	**2.3**	ND (1)	–	ND (1)	ND (1)
	08-27-93	7.6	ND (1.0)	ND (1.0)	ND (1.0)	–	ND (1.0)	ND (1.0)
	10-29-96	7.4	ND (1.0)	ND (1.0)	ND (1.0)	–	ND (1.0)	ND (1.0)
	01-28-97	7.2	ND (1.0)	**0.69 J**	ND (1.0)	–	ND (1.0)	ND (1.0)
	04-29-97	7.0	**0.33 J**	**0.79 J**	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)
	07-22-97	7.4	ND (1.0)	**0.32 J**	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)
	10-14-97	–	**0.24**	**0.57**	ND (0.2)	ND (0.2)	ND (0.2)	ND (0.2)
	01-19-98	7.5	ND (0.2)	ND (0.5)	ND (0.2)	ND (0.2)	ND (0.2)	ND (1.0)
	04-13-98	7.3	**0.56**	ND (0.5)	ND (0.2)	ND (0.2)	ND (0.2)	ND (1.0)
	07-27-98	6.9	ND (0.20)	**1.2**	ND (0.20)	ND (0.20)	ND (0.20)	ND (1.0)
	10-19-98	6.5	ND (0.20)	**0.97**	ND (0.20)	ND (0.20)	ND (0.20)	ND (1.0)
6-S-3	02-13-95	7.3	ND (2.5)	**2.0 J**	ND (2.5)	–	ND (2.5)	ND (2.5)
	09-21-95	6.9	ND (5)	ND (5)	ND (5)	–	ND (5)	ND (5)
	12-07-95	6.8	ND (5)	ND (5)	ND (5)	–	ND (5)	**4.5 J**
	03-26-96	7.0	ND (5)	**3.7 J**	ND (5)	–	ND (5)	ND (5)
	10-31-96	6.8	**2.4**	**7.2**	ND (2.0)	–	ND (2.0)	ND (2.0)
	01-28-97	7.0	ND (2.0)	**2**	ND (2.0)	–	ND (2.0)	ND (2.0)
	04-29-97	6.9	ND (2.5)	**1.5 J**	ND (2.5)	ND (2.5)	ND (2.5)	ND (2.5)
	07-22-97	7.1	ND (2.5)	**1.7 J**	ND (2.5)	ND (2.5)	ND (2.5)	ND (2.5)
	10-14-97	7.3	**0.27**	**2.6**	ND (0.20)	ND (0.20)	ND (0.20)	**0.79**
	04-13-98	l6.9	ND (0.20)	**3.2**	ND (0.20)	ND (0.20)	ND (0.20)	ND (1.0)
	10-19-98	6.1	ND (0.20)	**3.9**	ND (0.20)	ND (0.20)	ND (0.20)	ND (1.0)
6-S-6	09-09-93	7.4	**470**	**5,300**	ND (250)	–	**460**	ND (250)
	02-14-95	7.6	**330**	**1,200**	ND (200)	–	**90 J**	ND (200)
	09-22-95	7.4	**230**	**1,700**	ND (100)	–	**80 J**	ND (100)
	12-07-95	7.2	**220**	**2,000**	ND (100)	–	**150**	ND (100)
	03-27-96	7.4	**880**	**3,000**	ND (100)	–	**340**	ND (100)
	10-30-96	7.3	**690**	**1,000**	ND (50)	–	**110**	ND (50)
	01-30-97	7.1	**800**	**730**	ND (25)	–	**95**	ND (25)
	01-30-97	7.1	**680**	**690**	ND (20)	–	**94**	ND (20)
	02-03-97	7.4	**770**	**640**	ND (50)	–	**97**	ND (50)
	02-03-97	7.4	**800**	**740**	ND (25)	–	**91**	ND (25)
	05-02-97	7.1	**540**	**290**	ND (25)	**80**	**40**	ND (25)
	07-23-97	7.2	**500**	**120**	ND (25)	**87**	**31**	ND (25)
	10-15-97	6.9	**400**	**130**	**1.3**	**77**	**24**	ND (0.2)
	10-15-97	6.9	**370**	**89**	**1.3**	**73**	**24**	ND (0.2)
	01-20-98	7.2	**350**	**210**	**0.96**	**75**	**31**	ND (1.0)
	04-14-98	6.9	**370**	**100**	**1.1**	**76**	**22**	ND (1.0)
	07-28-98	6.7	**290**	**160**	**2.7**	**50**	**23**	ND (1.0)
	07-28-98	6.7	**260**	**150**	**3.2**	**47**	**23**	ND (1.0)

Table 4. Cumulative summary of chlorinated volatile organic compounds (VOC) data collected at Area 6 Naval Air Station Whidbey Island extraction wells and monitoring wells (from Foster Wheeler Environmental, 1999)—*Continued*

[Results are reported in micrograms per liter (μg/L); Effluent limitations are shown on this table for comparison to ground-water quality criteria; 1995 and later samples analyzed by U.S. Environmental Protection Agency Method 601; 1993 samples analyzed by U.S. Environmental Protection Agency Method 5242; duplicate samples for a given well are labeled with the same date; **TCE**, trichloroethene; **TCA**, 1,1,1-trichloroethane; **DCA**, 1,1-dichloroethane; ***cis*-DCE**, *cis*-1,2-dichloroethene; **DCE**, 1,1,-dicloroethene; **VC**, vinyl chloride; ND ( ), indicates parameter not detected with detection limit in parenthesis; NR, no reading; –, not analyzed for indicated parameter; J, estimated value—detected, but below quantitation limit; **bold** indicates detected values]

Well identification number	Sample date	Parameters						
		pH	TCE	TCA	DCA	*cis*-DCE	DCE	VC
	10-23-98	6.4	**340**	**57**	**1.2**	**47**	**15**	ND (1.0)
	10-23-98	6.4	**340**	**55**	**1.2**	**45**	**15**	ND (1.0)
6-S-7	10-29-96	7.0	ND (1.0)	ND (1.0)	ND (1.0)	–	ND (1.0)	ND (1.0)
	02-03-97	6.9	**1.4**	**2**	ND (1.0)	–	ND (1.0)	ND (1.0)
	04-30-97	6.0	ND (1.0)	**0.27 J**	ND (1.0)	–	ND (1.0)	ND (1.0)
	07-28-97	7.2	**0.95 J**	**1.2**	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)
	10-17-97	6.8	**1.2**	**2.8**	ND (0.2)	ND (0.2)	ND (0.2)	ND (1.0)
	10-19-98	6.1	ND (0.20)	ND (0.50)	ND (0.20)	ND (0.20)	ND (0.20)	ND (1.0)
6-S-10	09-11-93	10.4	ND (5)	**21**	ND (5)	–	ND (5)	ND (5)
	02-15-95	8.1	ND (1)	**3.7**	ND (1)	–	ND (1)	ND (1)
	09-22-95	7.5	ND (1)	**5.2**	ND (1)	–	ND (1)	ND (1)
	12-09-95	7.9	**1.5**	**11**	ND (1)	–	ND (1)	ND (1)
	03-28-96	7.4	**4.4**	**20**	**0.21 J**	–	**0.90 J**	ND (1)
	11-02-96	7.8	**1.0**	**2.7**	ND (1.0)	–	ND (1.0)	ND (1.0)
	02-03-97	7.6	**0.81 J**	**1.2**	ND (1.0)	–	ND (1.0)	ND (1.0)
	05-01-97	7.6	ND (1.0)	**1.6**	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)
	07-28-97	8.1	ND (1.0)	**0.28 J**	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)
	10-28-97	7.3	ND (0.20)	ND (0.50)	ND (0.20)	ND (0.20)	ND (0.20)	ND (1.0)
	04-13-98	7.7	ND (0.20)	ND (0.50)	ND (0.20)	ND (0.20)	ND (0.20)	ND (1.0)
	10-19-98	6.7	ND (0.20)	ND (0.50)	ND (0.20)	ND (0.20)	ND (0.20)	ND (1.0)
6-S-11	09-13-93	7.3	ND (1)	ND (1)	ND (1)	–	ND (1)	ND (1)
	02-15-95	7.2	ND (1)	ND (1)	ND (1)	–	ND (1)	ND (1)
	09-26-95	7.1	ND (1)	**1.3**	ND (1)	–	ND (1)	ND (1)
	12-09-95	7.1	**1.4**	**8.5**	ND (1)	–	ND (1)	ND (1)
	03-27-96	7.1	ND (1)	**1.8**	ND (1)	–	ND (1)	ND (1)
	11-02-96	NR	ND (1.0)	ND (1.0)	ND (1.0)	–	ND (1.0)	ND (1.0)
	02-03-97	6.8	**1**	**1.3**	ND (1.0)	–	ND (1.0)	ND (1.0)
	04-30-97	6.8	ND (1.0)	**7.9**	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)
	07-24-97	7.0	**0.98 J**	**0.27 J**	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)
	10-16-97	7.1	**1.8**	**2.3**	ND (0.2)	ND (0.2)	ND (0.2)	ND (0.2)
	01-19-98	7.4	ND (0.2)	ND (0.5)	ND (0.2)	ND (0.2)	ND (0.2)	ND (1.0)
	04-14-98	7.1	ND (0.2)	ND (0.5)	ND (0.2)	ND (0.2)	ND (0.2)	ND (1.0)
	07-27-98	6.9	ND (0.20)	ND (0.50)	ND (0.20)	ND (0.20)	ND (0.20)	ND (1.0)
	10-23-98	6.5	ND (0.20)	ND (0.50)	ND (0.20)	ND (0.20)	ND (0.20)	ND (1.0)
6-S-12	12-08-95	7.2	**1.4**	**7.2**	ND (1)	–	ND (1)	ND (1)
	10-31-96	7.1	**1.9**	**5.2**	ND (1.0)	–	ND (1.0)	ND (1.0)
	01-28-97	6.8	**1.2**	**1.9**	ND (1.0)	–	ND (1.0)	ND (1.0)
	05-01-97	6.7	ND (1.0)	1.5	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)
	07-24-97	7.0	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)
	10-16-97	6.7	**0.7**	**1.6**	ND (0.20)	ND (0.20)	ND (0.20)	ND (1.0)
	10-20-98	6.5	ND (0.20)	ND (0.50)	ND (0.20)	ND (0.20)	ND (0.20)	ND (1.0)
6-S-13	09-13-93	7.4	–	–	–	–	–	–
	02-13-95	7.8	ND (25)	**340**	**41**	–	ND (25)	ND (25)
	09-25-95	7.8	ND (25)	**660**	**310**	–	**25**	ND (25)

Table 4. Cumulative summary of chlorinated volatile organic compounds (VOC) data collected at Area 6 Naval Air Station Whidbey Island extraction wells and monitoring wells (from Foster Wheeler Environmental, 1999)—*Continued*

[Results are reported in micrograms per liter (µg/L); Effluent limitations are shown on this table for comparison to ground-water quality criteria; 1995 and later samples analyzed by U.S. Environmental Protection Agency Method 601; 1993 samples analyzed by U.S. Environmental Protection Agency Method 5242; duplicate samples for a given well are labeled with the same date; **TCE**, trichloroethene; **TCA**, 1,1,1-trichloroethane; **DCA**, 1,1-dichloroethane; ***cis*-DCE**, *cis*-1,2-dichloroethene; **DCE**, 1,1,-dicloroethene; **VC**, vinyl chloride; ND ( ), indicates parameter not detected with detection limit in parenthesis; NR, no reading; –, not analyzed for indicated parameter; J, estimated value—detected, but below quantitation limit; **bold** indicates detected values]

Well identification number	Sample date	Parameters						
		pH	TCE	TCA	DCA	*cis*-DCE	DCE	VC
	12-08-95	7.4	**12 J**	**370**	**580**	–	**36 J**	ND (40)
	03-27-96	8.0	ND (25)	**160**	**460**	–	**10 J**	ND (25)
	10-30-96	7.8	ND (20)	**210**	**250**	–	**12 J**	ND (20)
	01-18-97	7.5	ND (20)	**210**	**510**	–	ND (20)	ND (20)
	04-30-97	8.0	ND (10)	**130**	**300**	ND (10)	**17**	ND (10)
	07-28-97	7.3	ND (10)	**65**	**210**	ND (10)	**7.9 J**	ND (10)
	10-15-97	7.4	**5.0**	**68**	**330**	**0.66**	**9.3**	**0.53**
	04-15-98	7.5	**1.8**	**54**	**240**	ND (0.20)	**14**	ND (1.0)
	07-28-98	7.2	**0.25**	**160**	**370**	ND (0.20)	**8.7**	ND (1.0)
	07-28-98	7.2	**0.25**	**140**	**360**	ND (0.20)	**9.5**	ND (1.0)
	10-26-98	6.8	ND (2.0)	**44**	**190**	ND (2.0)	**5.7**	ND (10)
	10-26-98	6.8	ND (2.0)	**51**	**190**	ND (2.0)	**5.9**	ND (10)
6-S-14	02-14-95	7.4	ND (1)	**4.8**	ND (1)	–	ND (1)	ND (1)
	09-24-95	6.7	ND (1)	ND (1)	ND (1)	–	ND (1)	ND (1)
	03-26-96	6.8	ND (1)	**0.64 J**	**0.72 J**	–	ND (1)	**1.2**
	10-15-97	6.8	**4.1**	**13**	**1.1**	**0.37**	**0.38**	**0.5**
	10-23-98	6.5	ND (0.20)	ND (0.50)	**1.1**	ND (0.20)	ND (0.20)	ND (1.0)
6-S-15	02-14-95	7.5	ND (1)	ND (1)	ND (1)	–	ND (1)	ND (1)
	09-24-95	7.3	ND (1)	**1.7**	ND (1)	–	ND (1)	ND (1)
	12-06-95	6.8	ND (1)	**0.75 J**	ND (1)	–	ND (1)	ND (1)
	07-25-97	7.0	ND (1)	ND (1)	ND (1)	ND (1)	ND (1)	ND (1)
	10-28-97	6.6	ND (0.20)	ND (0.50)	ND (0.20)	ND (0.20)	ND (0.20)	ND (1.0)
	10-21-98	6.5	ND (0.20)	ND (0.50)	ND (0.20)	ND (0.20)	ND (0.20)	ND (1.0)
6-S-19	02-13-95	7.1	ND (2)	**7.7**	ND (2)	–	ND (2)	**2.5**
	09-21-95	6.8	ND (5)	ND (5)	ND (5)	–	ND (5)	ND (5)
	12-07-95	6.8	ND (5)	**6.7**	ND (5)	–	ND (5)	**4.2 J**
	03-29-96	6.8	**6.6**	**42**	ND (2)	–	**1.5 J**	ND (2)
	08-27-96	6.7	ND (1.0)	**3.9**	ND (1.0)	–	ND (1.0)	ND (1.0)
	10-31-96	7.1	**4.7**	**20**	ND (2.0)	–	**0.37 J**	ND (2.0)
	01-28-97	6.9	ND (2.0)	**7.1**	ND (2.0)	–	ND (2.0)	ND (2.0)
	04-29-97	7.0	ND (2.5)	**5.1**	ND (2.5)	ND (2.5)	ND (2.5)	**0.99 J**
	07-23-97	6.9	ND (2.5)	**3.4**	ND (2.5)	ND (2.5)	ND (2.5)	ND (2.5)
	10-14-97	7.2	ND (0.2)	**5.4**	ND (0.2)	ND (0.2)	**1.6**	**2.0**
	01-19-98	7.0	ND (0.2)	**8.1**	ND (0.2)	ND (0.2)	ND (0.2)	ND (1.0)
	04-13-98	6.7	ND (0.2)	**8.4**	ND (0.2)	ND (0.2)	ND (0.2)	**1.4**
	07-27-98	6.6	ND (0.20)	**8.7**	ND (0.20)	ND (0.20)	ND (0.20)	ND (1.0)
	10-22-98	6.3	ND (0.20)	**9.1**	ND (0.20)	ND (0.20)	ND (0.20)	**1.3**
	10-22-98	6.3	ND (0.20)	**9.3**	ND (0.20)	ND (0.20)	ND (0.20)	**1.3**
6-S-21	02-13-95	7.6	ND (50)	**810**	ND (50)	–	**29 J**	ND (50)
	09-21-95	8.1	ND (50)	**920**	ND (50)	–	**32 J**	ND (50)
	12-10-95	7.1	ND (25)	**360**	ND (25)	–	**17 J**	ND (25)
	03-27-96	7.6	ND (10)	**350**	ND (10)	–	**11**	ND (10)
	08-27-96	7.0	**2.8 J**	**260**	ND (10)	–	**16**	ND (10)

Table 4. Cumulative summary of chlorinated volatile organic compounds (VOC) data collected at Area 6 Naval Air Station Whidbey Island extraction wells and monitoring wells (from Foster Wheeler Environmental, 1999)—*Continued*

[Results are reported in micrograms per liter (µg/L); Effluent limitations are shown on this table for comparison to ground-water quality criteria; 1995 and later samples analyzed by U.S. Environmental Protection Agency Method 601; 1993 samples analyzed by U.S. Environmental Protection Agency Method 5242; duplicate samples for a given well are labeled with the same date; **TCE**, trichloroethene; **TCA**, 1,1,1-trichloroethane; **DCA**, 1,1-dichloroethane; ***cis*-DCE**, *cis*-1,2-dichloroethene; **DCE**, 1,1,-dicloroethene; **VC**, vinyl chloride; ND ( ), indicates parameter not detected with detection limit in parenthesis; NR, no reading; –, not analyzed for indicated parameter; J, estimated value—detected, but below quantitation limit; **bold** indicates detected values]

Well identification number	Sample date	Parameters						
		pH	TCE	TCA	DCA	*cis*-DCE	DCE	VC
	10-30-96	7.3	ND (10)	**240**	ND (10)	–	**7.9 J**	ND (10)
	01-31-97	7.2	ND (10)	**360**	ND (10)	–	**12**	ND (10)
	02-03-97	7.1	ND (10)	**340**	ND (10)	–	**10**	ND (10)
	01-30-97	7.1	ND (10)	**160**	ND (10)	–	**8.7 J**	ND (10)
	05-02-97	6.8	ND (10)	**290**	ND (10)	ND (10)	**14**	ND (10)
	07-23-97	7.3	ND (10)	**240**	ND (10)	ND (10)	**15**	ND (10)
	10-14-97	7.5	**0.26**	**310**	**0.64**	ND (0.2)	**20**	ND (0.2)
	10-14-97	7.5	**0.28**	**330**	**0.64**	ND (0.2)	**20**	ND (0.2)
	01-20-98	7.3	ND (0.2)	**360**	ND (0.2)	ND (0.2)	**21**	ND (1.0)
	01-20-98	7.3	ND (0.2)	**340**	ND (0.2)	ND (0.2)	**20**	ND (1.0)
	04-14-98	7.1	ND (0.2)	**380**	ND (0.2)	ND (0.2)	**18**	ND (1.0)
	07-29-98	6.9	**2.1**	**300**	**0.25**	ND (0.20)	**17**	ND (1.0)
	10-26-98	6.7	ND (0.20)	**310**	ND (0.20)	ND (0.20)	**11**	ND (1.0)
6-S-22	02-14-95	7.9	**5.8**	**14**	ND (1)	–	**1.3**	ND (1)
	09-25-95	8.1	ND (1)	**8.7**	**0.53 J**	–	ND (1)	ND (1)
	09-25-95	8.1	ND (1)	**9.1**	**0.57 J**	–	ND (1)	ND (1)
	12-11-95	7.7	**0.93 J**	**5.2**	ND (1)	–	ND (1)	ND (1)
	03-28-96	7.6	**1.7**	ND (1)	**0.56 J**	–	ND (1)	ND (1)
	11-05-96	6.9	ND (1.0)	**0.91 J**	ND (1.0)	–	ND (1.0)	ND (1.0)
	11-05-96	6.9	ND (1.0)	**0.73 J**	ND (1.0)	–	ND (1.0)	ND (1.0)
	01-30-97	7.0	**3.6**	**1.6**	ND (1.0)	–	ND (1.0)	ND (1.0)
	05-03-97	6.5	**5.1**	**1.7**	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)
	07-25-97	6.8	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)
	10-16-97	6.6	**0.44**	**0.87**	ND (0.20)	ND (0.20)	ND (0.20)	ND (1.0)
	10-21-98	6.7	ND (0.20)	ND (0.50)	ND (0.20)	ND (0.20)	ND (0.20)	ND (1.0)
6-S-23	09-10-93	7.5	ND (1)	**4.4**	**5.8**	–	ND (1)	**2.4**
	02-14-95	7.2	ND (1)	**3.6**	**1.1**	–	ND (1)	**0.47 J**
	09-24-95	6.6	**5.8**	**32**	**3.4**	–	**2.2 J**	**2.6**
	03-26-96	6.7	ND (1)	**2.6**	**4.6**	–	ND (1)	**1.7**
	10-15-97	6.6	**7.7**	**38**	**4.2**	**0.77**	**1.0**	**0.42**
6-S-23	04-14-98	6.6	ND (0.20)	ND (0.50)	**6.7**	ND (0.20)	ND (0.20)	**1.1**
	10-23-98	6.0	ND (0.20)	**1.8**	**5.3**	ND (0.20)	ND (0.20)	ND (1.0)
6-S-24	09-13-93	7.7	ND (1)	ND (1)	ND (1)	–	ND (1)	ND (1)
	02-15-95	8.1	**3.3 J**	**76**	ND (5)	–	**8.2**	ND (5)
	09-26-95	8.1	**2.0**	**64**	ND (2)	–	ND (2)	ND (2)
	12-08-95	7.4	**21.0**	**100**	ND (5)	–	**6.4**	ND (5)
	03-29-96	7.9	**5.1**	**34**	**0.80 J**	–	**0.66 J**	ND (1)
	08-17-96	8.2	ND (4.0)	**67**	ND (4.0)	–	**0.88 J**	ND (4.0)
	10-30-96	7.8	**1.6 J**	**120**	ND (5.0)	–	ND (5.0)	ND (5.0)
	01-28-97	7.5	**1.3 J**	**140**	ND (5.0)	–	ND (5.0)	ND (5.0)
	05-01-97	7.6	ND (10)	**270**	ND (10)	ND (10)	ND (10)	ND (10)
	07-24-97	7.7	ND (10)	**260**	ND (10)	ND (10)	ND (10)	ND (10)

Table 4. Cumulative summary of chlorinated volatile organic compounds (VOC) data collected at Area 6 Naval Air Station Whidbey Island extraction wells and monitoring wells (from Foster Wheeler Environmental, 1999)—*Continued*

[Results are reported in micrograms per liter (μg/L); Effluent limitations are shown on this table for comparison to ground-water quality criteria; 1995 and later samples analyzed by U.S. Environmental Protection Agency Method 601; 1993 samples analyzed by U.S. Environmental Protection Agency Method 5242; duplicate samples for a given well are labeled with the same date; **TCE**, trichloroethene; **TCA**, 1,1,1-trichloroethane; **DCA**, 1,1-dichloroethane; ***cis*-DCE**, *cis*-1,2-dichloroethene; **DCE**, 1,1,-dicloroethene; **VC**, vinyl chloride; ND ( ), indicates parameter not detected with detection limit in parenthesis; NR, no reading; –, not analyzed for indicated parameter; J, estimated value—detected, but below quantitation limit; **bold** indicates detected values]

Well identification number	Sample date	Parameters						
		pH	TCE	TCA	DCA	*cis*-DCE	DCE	VC
	10-16-97	7.6	**1.6**	**230**	ND (0.2)	ND (0.2)	**22.0**	ND (0.2)
	01-20-98	8.0	ND (0.2)	**98**	ND (0.2)	ND (0.2)	**0.4**	ND (1.0)
	04-15-98	7.6	**5.9**	**29**	ND (0.2)	**0.55**	**0.87**	ND (1.0)
	07-28-98	7.3	**1.1**	**26**	ND (0.20)	ND (0.20)	ND (0.20)	ND (1.0)
	10-26-98	6.7	**3.4**	**17**	ND (0.20)	ND (0.20)	ND (0.20)	ND (1.0)
6-S-25	09-09-93	6.8	ND (200)	**3,200**	ND (200)	–	ND (200)	ND (200)
	02-13-95	8.0	ND (100)	**2,500**	ND (100)	–	**26 J**	ND (100)
	09-25-95	8.4	ND (100)	**2,400**	ND (100)	–	ND (100)	ND (100)
	12-08-95	7.4	**19 J**	**1,600**	ND (50)	–	**56**	ND (50)
	03-27-96	9.0	ND (100)	**1,400**	ND (100)	–	ND (100)	ND (100)
	08-17-96	8.2	ND (200)	**3,100**	ND (200)	–	**67 J**	ND (200)
	08-17-96	8.2	ND (200)	**3,200**	ND (200)	–	ND (200)	ND (200)
	10-30-96	8.0	ND (200)	**2,300**	ND (200)	–	ND (200)	ND (200)
	01-28-97	7.8	ND (200)	**3,500**	ND (200)	–	ND (200)	ND (200)
	04-30-97	8.2	ND (100)	**2,700**	ND (100)	ND (100)	ND (100)	ND (100)
	07-28-97	7.8	ND (100)	**3,300**	ND (100)	ND (100)	ND (100)	ND (100)
	10-15-97	–	**15**	**2,200**	**3.7**	**1.6**	**25**	ND (0.2)
	01-20-98	8.0	**1.1**	**4,300**	**2.1**	ND (0.2)	**54**	ND (1.0)
	04-15-98	7.7	**3.3**	**810**	**7.9**	ND (0.2)	**35**	ND (1.0)
	07-28-98	7.4	**0.51**	**5,100**	**6.7**	ND (0.20)	**67**	ND (1.0)
	10-26-98	7.1	**2.1**	**3,600**	**4.3**	ND (0.20)	**50**	ND (1.0)
	10-26-98	7.1	**2.3**	**3,800**	**4.8**	ND (0.20)	**50**	ND (1.0)
6-S-26	11-03-96	7.3	ND (1.0)	**0.48 J**	ND (1.0)	–	ND (1.0)	ND (1.0)
	02-03-97	7.4	**0.28 J**	**0.46 J**	ND (1.0)	–	ND (1.0)	ND (1.0)
	05-01-97	7.4	ND (1.0)	**0.69 J**	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)
	07-28-97	7.8	**2.0**	**1.0**	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)
	10-17-97	7.4	**0.9**	**1.4**	ND (0.20)	ND (0.20)	ND (0.20)	ND (1.0)
	10-19-98	6.3	ND (0.20)	ND (0.50)	ND (0.20)	ND (0.20)	ND (0.20)	ND (1.0)
6-S-27	09-16-96	8.3	ND (10)	**140**	ND (10)	–	**4.7 J**	ND (10)
	10-31-96	8.0	ND (5.0)	**73**	ND (5.0)	–	**3.2 J**	ND (5.0)
	01-28-97	7.8	ND (5.0)	**42**	ND (5.0)	–	ND (5.0)	ND (5.0)
	05-02-97	7.8	ND (2.5)	**34**	ND (2.5)	ND (2.5)	**2.7**	ND (2.5)
	07-24-97	7.7	**1.2**	**20**	ND (2.5)	ND (2.5)	**1.9**	ND (2.5)
	10-27-97	8.1	**1.0**	**25**	ND (0.20)	ND (0.20)	**3.7**	ND (0.2)
	01-20-98	8.1	ND (0.2)	**18**	ND (0.20)	ND (0.20)	**2.1**	ND (1.0)
	04-16-98	7.8	**4.2**	**19**	ND (0.20)	ND (0.20)	**2.2**	ND (1.0)
	07-27-98	7.5	ND (0.20)	**17**	ND (0.20)	ND (0.20)	**1.4**	ND (1.0)
	10-23-98	7.2	ND (0.20)	**9.8**	ND (0.20)	ND (0.20)	**1.2**	ND (1.0)
6-S-28	09-17-96	8.1	ND (1.0)	ND (1.0)	ND (1.0)	–	ND (1.0)	ND (1.0)
	10-31-96	7.6	**1.4**	**3.3**	ND (1.0)	–	ND (1.0)	ND (1.0)
	01-28-97	7.3	**12**	**11**	ND (1.0)	–	**0.32 J**	ND (1.0)
	05-02-97	7.2	ND (1.0)	**0.51 J**	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)
	07-24-97	6.9	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)

Table 4. Cumulative summary of chlorinated volatile organic compounds (VOC) data collected at Area 6 Naval Air Station Whidbey Island extraction wells and monitoring wells (from Foster Wheeler Environmental, 1999)—*Continued*

[Results are reported in micrograms per liter (µg/L); Effluent limitations are shown on this table for comparison to ground-water quality criteria; 1995 and later samples analyzed by U.S. Environmental Protection Agency Method 601; 1993 samples analyzed by U.S. Environmental Protection Agency Method 5242; duplicate samples for a given well are labeled with the same date; **TCE**, trichloroethene; **TCA**, 1,1,1-trichloroethane; **DCA**, 1,1-dichloroethane; ***cis*-DCE**, *cis*-1,2-dichloroethene; **DCE**, 1,1,-dicloroethene; **VC**, vinyl chloride; ND ( ), indicates parameter not detected with detection limit in parenthesis; NR, no reading; –, not analyzed for indicated parameter; J, estimated value—detected, but below quantitation limit; **bold** indicates detected values]

Well identification number	Sample date	Parameters						
		pH	TCE	TCA	DCA	*cis*-DCE	DCE	VC
	10-27-97	7.5	**0.21**	**1.1**	ND (0.2)	ND (0.2)	ND (0.2)	ND (0.2)
	04-14-98	7.2	ND (0.20)	ND (0.50)	ND (0.20)	ND (0.20)	ND (0.20)	ND (1.0)
	10-20-98	6.5	ND (0.20)	ND (0.50)	ND (0.20)	ND (0.20)	ND (0.20)	ND (1.0)
6-S-29	05-02-97	6.7	**0.63 J**	**7.8**	**2.2**	ND (1.0)	ND (1.0)	**5.3**
	07-22-97	6.9	ND (5.0)	**5.5**	ND (5.0)	ND (5.0)	ND (5.0)	**4.9 J**
	10-28-97	6.6	ND (0.2)	**7.2**	**3.1**	ND (0.2)	ND (0.2)	**4.4**
	10-28-97	6.6	ND (0.2)	**6.0**	**2.5**	ND (0.2)	ND (0.2)	**3.3**
	01-19-98	7.0	ND (0.2)	**8.1**	**3.4**	ND (0.2)	**0.20**	**5.2**
	04-13-98	6.8	ND (0.2)	**7.1**	**4.1**	ND (0.2)	ND (0.2)	**5.6**
	07-28-98	6.5	ND (0.20)	**7.7**	**3.6**	ND (0.20)	ND (0.20)	**4.2**
	10-22-98	6.4	ND (0.20)	**8.3**	**4.2**	ND (0.20)	ND (0.20)	**5.5**
	10-22-98	6.4	ND (0.20)	**8.0**	**3.9**	ND (0.20)	ND (0.20)	**5.3**
MW-5	09-13-93	7.6	ND (1)	**1.1**	ND (1)	–	ND (1)	ND (1)
	02-15-95	7.5	ND (2)	**0.42 J**	ND (2)	–	ND (2)	ND (2)
	09-27-95	7.6	ND (1)	ND (1)	ND (1)	–	ND (1)	ND (1)
	12-11-95	7.5	ND (2)	ND (2)	ND (2)	–	ND (2)	ND (2)
	03-28-96	7.7	ND (1)	ND (1)	ND (1)	–	ND (1)	ND (1)
	08-16-96	7.8	ND (1.0)	ND (1.0)	ND (1.0)	–	ND (1.0)	ND (1.0)
	11-01-96	7.5	ND (1.0)	**0.43 J**	ND (1.0)	–	ND (1.0)	ND (1.0)
	01-29-97	7.5	ND (1.0)	**0.24 J**	ND (1.0)	–	ND (1.0)	ND (1.0)
	05-02-97	7.2	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)
	07-24-97	7.2	ND (1.0)	**0.40 J**	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)
	10-27-97	7.7	**0.33**	**1.2**	ND (0.20)	ND (0.20)	ND (0.20)	ND (1.0)
	04-14-98	7.3	ND (0.20)	**4.1**	ND (0.20)	ND (0.20)	ND (0.20)	ND (1.0)
	10-20-98	6.7	ND (0.20)	**5.6**	**0.23**	ND (0.20)	ND (0.20)	ND (1.0)
MW-6	03-28-96	7.8	ND (1)	ND (1)	ND (1)	–	ND (1)	ND (1)
	08-16-96	8.0	ND (1.0)	ND (1.0)	ND (1.0)	–	ND (1.0)	ND (1.0)
	10-31-96	7.6	ND (1.0)	ND (1.0)	ND (1.0)	–	ND (1.0)	ND (1.0)
	01-29-97	7.9	ND (1.0)	ND (1.0)	ND (1.0)	–	ND (1.0)	ND (1.0)
	05-02-97	NR	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)
	07-24-97	7.6	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)
	10-27-97	8.4	**0.53**	**0.71**	ND (0.20)	ND (0.20)	ND (0.20)	ND (1.0)
	10-20-98	6.9	ND (0.20)	**0.58**	ND (0.20)	ND (0.20)	ND (0.20)	ND (1.0)
MW-7	02-14-95	7.5	**350 J**	**3,500**	ND (500)	–	ND (500)	ND (500)
	02-14-95	7.5	**180 J**	**2,400**	ND (500)	–	**350 J**	ND (500)
	09-22-95	7.8	**390**	**3,700**	ND (200)	–	**330**	ND (200)
	12-08-95	7.2	**520**	**5,200**	ND (500)	–	ND (500)	ND (500)
	03-28-96	7.1	**480**	**5,300**	**160 J**	–	**380**	ND (200)
	03-28-96	7.1	**520**	**5,400**	**150 J**	–	**370**	ND (200)
	08-27-96	7.0	**670**	**3,500**	**110**	–	**270**	ND (100)
	10-31-96	7.5	**720**	**3,300**	**37 J**	–	**190 J**	ND (200)
	01-29-97	6.8	**1,000**	**1,600**	**150**	–	**120**	ND (50)
	05-02-97	7.0	**660**	**350**	**63**	**290**	**51**	ND (50)

Table 4. Cumulative summary of chlorinated volatile organic compounds (VOC) data collected at Area 6 Naval Air Station Whidbey Island extraction wells and monitoring wells (from Foster Wheeler Environmental, 1999)—*Continued*

[Results are reported in micrograms per liter (µg/L); Effluent limitations are shown on this table for comparison to ground-water quality criteria; 1995 and later samples analyzed by U.S. Environmental Protection Agency Method 601; 1993 samples analyzed by U.S. Environmental Protection Agency Method 5242; duplicate samples for a given well are labeled with the same date; **TCE**, trichloroethene; **TCA**, 1,1,1-trichloroethane; **DCA**, 1,1-dichloroethane; ***cis*-DCE**, *cis*-1,2-dichloroethene; **DCE**, 1,1,-dicloroethene; **VC**, vinyl chloride; ND ( ), indicates parameter not detected with detection limit in parenthesis; NR, no reading; –, not analyzed for indicated parameter; J, estimated value—detected, but below quantitation limit; **bold** indicates detected values]

Well identification number	Sample date	Parameters						
		pH	TCE	TCA	DCA	*cis*-DCE	DCE	VC
	05-02-97	7.0	**790**	**450**	ND (50)	**330**	**67**	ND (50)
	07-28-97	6.8	**530**	**220**	**65**	**230**	**49**	ND (20)
MW-7	07-28-97	6.8	**800**	**290**	**67**	**270**	**57**	ND (20)
	10-14-97	6.9	**420**	**140**	**49**	**160**	**32**	**0.28**
	01-20-98	7.1	**440**	**230**	**75**	**180**	**48**	ND (1.0)
	01-20-98	7.1	**400**	**190**	**71**	**160**	**40**	ND (1.0)
	04-15-98	6.9	**280**	**160**	**48**	**110**	**23**	ND (1.0)
	04-15-98	6.9	**380**	**330**	**85**	**170**	**44**	ND (1.0)
	07-29-98	6.5	**450**	**600**	**58**	**150**	**41**	ND (1.0)
	10-27-98	6.4	**510**	**460**	**63**	**180**	**51**	ND (1.0)
MW-8	02-14-95	7.2	ND (2.5)	**5.1**	ND (2.5)	–	ND (2.5)	ND (2.5)
	09-24-95	7.0	ND (5)	**3.7 J**	ND (5)	–	ND (5)	ND (5)
	12-09-95	6.9	**1.2 J**	**13**	ND (2)	–	ND (2)	**0.89 J**
	12-09-95	6.9	**1.9 J**	**14**	**3.6 J**	–	ND (5)	ND (5)
	03-28-96	7.3	ND (2)	**3.7**	**0.72 J**	–	ND (2)	**0.90 J**
	10-30-96	6.0	**6.2**	**8.8**	**0.54 J**	–	**0.55 J**	ND (2)
	01-29-97	6.8	ND (2.0)	**4.4**	**0.62 J**	–	ND (2.0)	ND (2.0)
	04-29-97	6.9	ND (2.0)	**2**	ND (2.0)	ND (2.0)	ND (2.0)	**0.60 J**
	07-25-97	6.9	ND (2.0)	**2.6**	ND (2.0)	ND (2.0)	ND (2.0)	**0.99 J**
	10-17-97	6.8	**0.5**	**6**	**1.3**	ND (0.2)	**0.55**	**1.1**
	10-22-98	6.4	ND (0.20)	**2.6**	**2.1**	ND (0.20)	ND (0.20)	**1.1**
MW-9	03-26-96	6.8	ND (2.5)	**7.8**	ND (2.5)	–	ND (2.5)	ND (2.5)
	02-03-97	7.0	**0.5 J**	**6.3**	ND (2.5)	–	ND (2.5)	ND (2.5)
	04-29-97	7.3	ND (5)	**5.7**	ND (5)	ND (5)	ND (5)	ND (5)
	07-25-97	6.8	ND (5)	**9.2**	ND (5)	ND (5)	ND (5)	ND (5)
	10-28-97	6.5	ND (0.20)	**13**	**0.65**	ND (0.20)	ND (0.20)	ND (0.2)
	10-22-98	6.6	ND (0.20)	**4.0**	**1.7**	ND (0.20)	ND (0.20)	ND (1.0)
MW-10	02-14-95	6.9	ND (2.5)	**11**	**21**	–	ND (2.5)	**11**
	09-24-95	6.6	ND (5)	ND (5)	**23**	–	ND (5)	**11**
	12-06-95	6.8	**1.4 J**	**2.5 J**	**31**	–	ND (5)	**20**
	03-26-96	6.8	ND (1)	**0.57 J**	**29**	–	ND (1)	ND (1)
	07-25-97	7.0	ND (2.0)	ND (2.0)	**18**	ND (2.0)	ND (2.0)	**3.4**
	10-29-97	6.6	ND (0.20)	ND (0.50)	**19**	**0.27**	ND (0.20)	**0.79**
	10-29-97	6.6	ND (0.20)	ND (0.50)	**19**	**0.27**	ND (0.20)	**0.81**
	04-14-98	6.8	ND (0.20)	ND (0.50)	**25**	**0.37**	ND (0.20)	**1.6**
	10-22-98	6.5	ND (0.20)	ND (0.50)	**17**	**0.23**	ND (0.20)	ND (1.0)
MW-11	09-24-95	7.6	**2.2**	**11**	ND (1)	–	**0.71 J**	ND (1)
	12-08-95	7.6	**3.2**	**25**	ND (1)	–	**1.6**	ND (1)
	03-26-96	7.2	ND (1)	**4.9**	ND (1)	–	ND (1)	ND (1)
	11-01-96	8.4	**1.7**	**9.1**	**1.4**	–	**0.21 J**	ND (1.0)
	01-29-97	7.6	**0.54 J**	**7.4**	**1.1**	–	ND (1.0)	ND (1.0)
	04-30-97	8.5	**0.38 J**	**6.1**	ND (1.0)	**0.96 J**	**0.72 J**	ND (1.0)
	07-22-97	7.8	**0.43 J**	**4.8**	**0.68 J**	**0.97 J**	ND (1.0)	ND (1.0)

Table 4. Cumulative summary of chlorinated volatile organic compounds (VOC) data collected at Area 6 Naval Air Station Whidbey Island extraction wells and monitoring wells (from Foster Wheeler Environmental, 1999)—*Continued*

[Results are reported in micrograms per liter (µg/L); Effluent limitations are shown on this table for comparison to ground-water quality criteria; 1995 and later samples analyzed by U.S. Environmental Protection Agency Method 601; 1993 samples analyzed by U.S. Environmental Protection Agency Method 5242; duplicate samples for a given well are labeled with the same date; **TCE**, trichloroethene; **TCA**, 1,1,1-trichloroethane; **DCA**, 1,1-dichloroethane; ***cis*-DCE**, *cis*-1,2-dichloroethene; **DCE**, 1,1,-dicloroethene; **VC**, vinyl chloride; ND ( ), indicates parameter not detected with detection limit in parenthesis; NR, no reading; –, not analyzed for indicated parameter; J, estimated value—detected, but below quantitation limit; **bold** indicates detected values]

Well identification number	Sample date	Parameters						
		pH	TCE	TCA	DCA	*cis*-DCE	DCE	VC
	10-28-97	7.2	**0.3**	**2.9**	**0.31**	**0.58**	ND (0.2)	ND (1.0)
	10-22-98	7.0	ND (0.20)	**2.0**	ND (0.20)	ND (0.20)	ND (0.20)	ND (1.0)
MW-14	03-26-96	7.2	ND (1)	**1.6**	ND (1)	–	ND (1)	ND (1)
	11-02-96	7.3	ND (1.0)	**2**	ND (1.0)	–	ND (1.0)	ND (1.0)
	01-29-97	7.2	ND (1.0)	**9.3**	ND (1.0)	–	ND (1.0)	ND (1.0)
	04-30-97	8.0	ND (1.0)	**0.72 J**	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)
	07-22-97	7.5	ND (1.0)	**0.65 J**	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)
	10-28-97	7.1	ND (0.20)	**0.62**	ND (0.20)	ND (0.20)	ND (0.20)	ND (1.0)
N637	09-11-93	7.5	**370**	**890**	ND (50)	–	ND (50)	ND (50)
	02-14-95	7.7	**450**	**930**	ND 100)	–	**29 J**	ND (100)
	09-26-95	7.2	**480**	**710**	ND (50)	–	**29 J**	ND (50)
	12-11-95	7.6	**280**	**240**	**2.8 J**	–	**16**	ND (10)
	03-28-96	7.1	**230**	**98**	ND (10)	–	**3.3 J**	ND (10)
	11-05-96	6.8	**150**	**29**	ND (1.0)	–	**2.3**	ND (1.0)
	01-30-97	7.1	**180**	**22**	ND (10)	–	ND (10)	ND (10)
	05-03-97	6.7	**100**	**7.2**	ND (5.0)	**1.9 J**	ND (5.0)	ND (5.0)
	07-25-97	6.4	**110**	**9.5**	ND (5.0)	ND (5.0)	ND (5.0)	ND (5.0)
	10-27-97	6.7	**100**	**17**	ND (0.2)	**6.2**	**2.1**	ND (0.2)
	01-20-98	6.6	**58**	**37**	ND (0.2)	**3.5**	**2.0**	ND (1.0)
	04-15-98	6.4	**62**	**58**	**1.5**	**6.4**	**3.1**	ND (1.0)
	04-15-98	6.4	**53**	**60**	**1.6**	**5.9**	**3.2**	ND (1.0)
	07-28-98	6.2	**61**	**23**	ND (0.20)	**3.7**	**0.77**	ND (1.0)
	07-28-98	6.2	**54**	**24**	ND (0.20)	**3.6**	**0.75**	ND (1.0)
	10-27-98	6.2	**120**	**73**	**2.1**	**9.7**	**4.3**	ND (1.0)
N638	03-29-96	8.2	ND (100)	**3,600**	ND (100)	–	**91 J**	ND (100)
	11-05-96	7.0	ND (100)	**1,400**	ND (100)	–	**46 J**	ND (100)
	01-30-97	7.1	**23**	**670**	ND (20)	–	**45**	ND (20)
	05-03-97	7.3	**8.4 J**	**380**	ND (20)	ND (20)	**17 J**	ND (20)
	07-25-97	6.9	**24.0**	**330**	ND (20)	ND (20)	**41**	ND (20)
	10-27-97	7.1	**38**	**680**	**1.8**	**13**	**87**	ND (0.2)
	10-27-97	7.1	**38**	**650**	**1.7**	**13**	**96**	ND (0.2)
	01-20-98	7.2	**19**	**250**	**0.32**	**1.6**	**32**	ND (1.0)
	04-15-98	6.9	**34**	**310**	**3.4**	**6.2**	**38**	ND (1.0)
	07-29-98	6.5	**27**	**540**	**1.6**	**1.6**	**42**	ND (1.0)
	10-27-98	6.7	**44**	**280**	**0.98**	**3.7**	**34**	ND (1.0)
	10-27-98	6.7	**46**	**290**	**1.1**	**4.0**	**34**	ND (1.0)
6-I-1	09-25-95	8.5	ND (1)	**1.9**	ND (1)	–	ND (1)	ND (1)
	12-11-95	8.0	**0.41 J**	**2.7**	ND (1)	–	ND (1)	ND (1)
	03-28-96	8.4	**0.93 J**	**3.0**	ND (1)	–	ND (1)	ND (1)
	11-05-96	8.4	**0.39 J**	**0.97 J**	ND (1.0)	–	ND (1.0)	ND (1.0)
	01-30-97	8.4	**2.6**	**2.8**	ND (1.0)	–	ND (1.0)	ND (1.0)

Table 4. Cumulative summary of chlorinated volatile organic compounds (VOC) data collected at Area 6 Naval Air Station Whidbey Island extraction wells and monitoring wells (from Foster Wheeler Environmental, 1999)—*Continued*

[Results are reported in micrograms per liter (µg/L); Effluent limitations are shown on this table for comparison to ground-water quality criteria; 1995 and later samples analyzed by U.S. Environmental Protection Agency Method 601; 1993 samples analyzed by U.S. Environmental Protection Agency Method 5242; duplicate samples for a given well are labeled with the same date; **TCE**, trichloroethene; **TCA**, 1,1,1-trichloroethane; **DCA**, 1,1-dichloroethane; ***cis*-DCE**, *cis*-1,2-dichloroethene; **DCE**, 1,1,-dicloroethene; **VC**, vinyl chloride; ND ( ), indicates parameter not detected with detection limit in parenthesis; NR, no reading; –, not analyzed for indicated parameter; J, estimated value—detected, but below quantitation limit; **bold** indicates detected values]

Well identification number	Sample date	Parameters						
		pH	TCE	TCA	DCA	*cis*-DCE	DCE	VC
	05-03-97	8.4	ND (1.0)	**0.66 J**	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)
	07-25-97	8.3	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)
	10-16-97	8.4	**0.36**	**0.62**	ND (0.20)	ND (0.20)	ND (0.20)	ND (1.0)
	10-21-98	7.8	ND (0.20)	ND (0.50)	ND (0.20)	ND (0.20)	ND (0.20)	ND (1.0)
6-I-3	02-14-95	7.9	ND (1)	**3.9**	ND (1)	–	ND (1)	ND (1)
	09-27-95	8.2	ND (1)	**0.82 J**	ND (1)	–	ND (1)	ND (1)
	12-07-95	7.2	ND (1)	**0.52 J**	ND (1)	–	ND (1)	ND (1)
	03-27-96	7.8	ND (1)	ND (1)	ND (1)	–	ND (1)	ND (1)
	10-30-96	8.0	ND (1.0)	ND (1.0)	ND (1.0)	–	ND (1.0)	ND (1.0)
	01-29-97	7.7	**4.0**	**4.7**	ND (1.0)	–	ND (1.0)	ND (1.0)
	05-02-97	7.6	ND (1.0)	**0.96 J**	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)
	07-23-97	7.8	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)
	10-15-97	7.6	**1.9**	**1.1**	ND (0.20)	**0.2**	**0.2**	ND (1.0)
	10-21-98	7.3	ND (0.20)	ND (0.50)	ND (0.20)	ND (0.20)	ND (0.20)	ND (1.0)
6-I-8	02-13-95	7.9	ND (1)	ND (1)	ND (1)	–	ND (1)	ND (1)
	09-21-95	8.1	ND (1)	**1.4**	ND (1)	–	ND (1)	ND (1)
	12-10-95	7.6	**0.38 J**	**3.0**	**0.48 J**	–	ND (1)	ND (1)
	03-27-96	7.9	ND (1)	ND (1)	ND (1)	–	ND (1)	ND (1)
	10-30-96	7.5	ND (1.0)	ND (1.0)	ND (1.0)	–	ND (1.0)	ND (1.0)
	01-31-97	7.6	**1.1**	**1.3**	ND (1.0)	–	ND (1.0)	ND (1.0)
	05-02-97	7.7	ND (1.0)	**0.54 J**	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)
	07-23-97	8.0	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)	ND (1.0)
	10-14-97	8.2	ND (0.20)	ND (0.50)	ND (0.20)	ND (0.20)	ND (0.20)	ND (0.2)
	10-21-98	7.3	ND (0.20)	ND (0.50)	ND (0.20)	ND (0.20)	ND (0.20)	ND (1.0)

The degradation products DCE and VC are the only two contaminants in the southern plume that have been measured at concentrations that routinely exceeded the site-specific cleanup levels of 0.07 and 0.02 µg/L (URS Consultants, 1993a), respectively. The data do not indicate that a substantial mass of DCE or VC is present. Rather, exceedances were common because of the comparatively low site-specific cleanup levels for the compounds (USEPA drinking water Maximum Contaminant Levels (MCL) are 7 and 2 µg/L, respectively). DCE was detected at concentrations up to 13 µg/L in the southern plume during 1991, but was not detected in 1995. It has been detected sporadically at the southern base boundary at concentrations up to 1.6 µg/L during 1995–98. VC was detected at up to 53.5 µg/L in the southern plume during 1991, and at lower concentrations in 1995. It has been detected consistently beneath and downgradient of the Area 6 landfill at less than 6 µg/L since 1996. Both DCE and VC concentrations appear to have decreased between 1991–95, but those trends are uncertain due to the sparse and inconsistent data. VC concentrations apparently decreased in many wells during 1995–98, but that trend is also not clear and consistent.

None of the VOC's of interest in the southern plume have been detected in the off-base wells AHI and MW-3B, with the exception of one detection of

TCE at 0.12 µg/L in MW3-B in September 1991. The site labeled AHI on figures 10–15 was actually two adjacent private wells that were sampled in 1991 and 1992, and have since been abandoned. Well MW-3B is a City of Oak Harbor monitoring well that has been sampled at least annually since 1990. The detection limits for all of those historical analyses were 1 µg/L or less. The lack of contamination in the downgradient, off-base wells suggests that VOC's found in ground water at the southern site boundary are not being transported in detectable concentrations to locations beyond the adjacent Oak Harbor landfill.

Some preliminary conclusions regarding the effectiveness of natural attenuation in the southern plume can be made based on changes in contaminant concentrations between 1991 and 1995. First, assuming that there was a substantial mass of TCE and TCA in the 2.2 million gallons of liquids and sludges disposed of in the Area 6 landfill, natural attenuation processes have been very effective at controlling the migration of that original contaminant mass. Essentially no TCE, and only low concentrations of TCA and intermediate degradation products have been observed in the southern plume. These data suggest that substantial biodegradation of both source materials and degradation products have occurred within and beneath the landfill. That conclusion is somewhat uncertain because the original mass of contaminants disposed of is not known. A second conclusion that can be made is that natural attenuation alone has prevented the southern plume from expanding much further than the southern site boundary. The VOC's found in the southern plume in 1991 had not been observed in the nearest off-base downgradient wells AHI or MW-3B before pump and treat began (1995) even though waste disposal in the Area 6 landfill began 26 years earlier. The uncertainty in that conclusion (which is addressed in the "Effectiveness of Natural Attenuation" section of this report) is whether enough time had passed to allow nonattenuated transport of contaminants to reach the off-base downgradient wells.

Western Contaminant Plume

In contrast to the southern plume, contaminant concentration trends in the western plume demonstrate limited degradation and substantial downgradient transport of contaminants from the original source area. TCA concentrations decreased beneath the source area during 1991–95, likely in response to a decrease in loading of TCA to ground water after liquid waste disposal stopped in the mid-1980's. However, TCA concentrations increased by an order of magnitude near the southern extent of the plume (wells 6-S-25 and 6-S-13) during the same time period. TCE concentrations also decreased beneath the source area during 1991–95 and increased near the leading edge of the plume (well 6-S-6), but the TCE plume had not expanded as far southward as the TCA plume. None of the VOC's of interest were detected during 1991-95 in the downgradient off-base wells MW-1, MW-5, and MW-6, with the exception of a few TCA detections of less than 1.5 µg/L in MW-5. Changes in concentrations of the TCA degradation product DCA followed a similar pattern to that observed for TCA, in that source area concentrations decreased and leading-edge concentrations increased. DCE concentrations appeared to have decreased throughout the plume between 1991 and 1995.

Some preliminary conclusions regarding the effectiveness of natural attenuation in the western plume can be made based on changes in contaminant concentrations between 1991 and 1995. In general, natural attenuation was not effective at controlling the on-base migration of VOC's during 1991-95, although little off-base migration of contaminants to the south of the site boundary was observed. Given the downgradient extent of the TCA plume observed in December 1992 (1,500 µg/L at well 6-S-25), it is possible that natural attenuation processes changed dramatically once the plume migrated beneath the Oak Harbor landfill, thus limiting further plume expansion. Uncertainties arise in that the off-base monitoring network may not have been extensive enough to observe plume migration, and enough time may not have passed to allow nonattenuated transport of contaminants to reach the downgradient wells. Those uncertainties are addressed in the "Effectiveness of Natural Attenuation" section of this report.

A second conclusion regarding natural attenuation in the western plume is that TCE migration appears to be more limited than TCA migration. Published sorption coefficients (K_{oc}) for TCA are generally slightly greater than those for TCE (Howard, 1993; Mackay and others, 1992), suggesting that more TCA should be sorbed to sediments and TCA migration should be more attenuated if biodegradation were not affecting the attenuation. The opposite case was observed for data between 1991 and 1995. The difference in extents in the TCA and TCE plumes may reflect different dates of disposal.

An additional observation from the contaminant distribution maps is that the specific source areas for TCA and TCE in the western plume are in different locations. The source area for TCE appears to be located where originally hypothesized (URS Consultants, 1993b), directly below the former hazardous waste storage area near wells PW-1 and N6-37. However, TCA concentrations increased during 1991-95 from 12 to 810 µg/L in well 6-S-21, which is up-gradient of the originally suspected source area near well N6-38. Those data suggest that a contaminant source may be located further north than the original hypothesized location. TCA concentrations in 6-S-21 have remained relatively stable at 200–400 µg/L during 1995-98, but TCA concentrations in N6-38 decreased from 3,600 to 346 µg/L during that same period. Soil vapor data collected in 1991 (URS Consultants, 1993b) show the highest TCA concentrations in an approximately 200-ft radius circle surrounding well 6-S-21.

Contaminant Degradation Indicated by Ground-Water Chemistry Data

Contaminant concentration changes over time provide only indirect evidence that destructive attenuation processes are occurring at Area 6. More direct evidence for contaminant degradation was ascertained from the 1998 ground-water chemistry data. The distribution of parent contaminant compounds relative to the distribution of intermediate degradation products provides evidence for the types, locations, and relative rates of degradation reactions that have occurred. The locations of inferred degradation can then be compared to ground-water redox conditions and to changes in concentrations of redox-sensitive chemicals along ground-water flow paths to further confirm inferred degradation processes.

Ground-water chemistry data primarily from 1998 were used to evaluate contaminant degradation at Area 6, because both redox and contaminant data are comprehensive for that year (tables 3 and 4). Although the 1998 data were somewhat influenced by remediation activities, it is thought that both the pump and treat system and the landfill cap have had only minimal impact on contaminant degradation and redox processes. The extraction wells may be mixing ground waters with different redox conditions due their high pumping rates and long screened-intervals, but the 1998 aquifer tests showed that their zones of influence extend outward only a few hundred feet. Treated water from the pump and treat system is discharged to the northward draining swale where most of it infiltrates near the northern site boundary. The treated water has DO concentrations of around 10 mg/L (similar to precipitation), but it is infiltrating over an area of the aquifer where shallow ground water is already aerobic, so the impact on ambient redox conditions is minimal. The landfill cap is not allowing oxygenated precipitation to recharge the aquifer, but the high organic carbon content of the landfill was consuming all available oxygen even before the cap was constructed. In addition, the landfill cap is now a permanent feature of site that will remain in place even if natural attenuation were to replace pump and treat as a remediation strategy.

Southern Contaminant Plume

Changes in contaminant concentrations over time indicated that natural attenuation has been effective at controlling the migration of contaminants in the southern plume, and ground-water chemistry data indicates that contaminant degradation in particular is an important attenuation process. The predominance of intermediate degradation products relative to parent compounds in the southern plume (table 5) is evidence for substantial reductive dechlorination of TCA and TCE. Although the absolute concentrations are low, the ratios TCA/DCA and TCE/*cis*DCE have been consistently low for wells beneath and immediately downgradient of the Area 6 landfill (table 5). These low ratios indicate that the majority of the TCA and TCE has been biodegraded through reductive dechlorination into di-chlorinated compounds. The ratios of DCE/VC and *cis*DCE/VC also are consistently low throughout the southern plume (except in well PW-7), which indicate that DCE compounds may also have been degraded through reductive dechlorination to VC. The predominance of intermediate degradation products in wells MW-10, PW-7, and PW-2 along a flow path through the southern plume also suggests reductive dechlorination (fig. 16). The ratio DCA/CA is not particularly diagnostic of reductive dechlorination of 1,1,-DCA because CA is rapidly degraded through abiotic processes, and thus not often detectable in ground water. CA was detected in well PW-9 at 1.0 µg/L, which demonstrates that it is being generated. The lack of detectable DCE (the intermediate product of abiotic dehydrohalogenation of TCA) in the southern plume is likely due to the relatively low TCA concentrations. Detectable DCE concentrations (greater than 0.2 µg/L) were generally not observed at Area 6 wells when TCA concentrations were less than about 10 µg/L.

The area beneath the Area 6 landfill where intermediate degradation products predominate over parent compounds coincides with the previously delineated strongly reducing redox zone (fig. 8). The sulfate-reducing and methanogenic conditions in that strongly reducing zone are ideal for reductive dechlorination of TCA, TCE, and their dichlorinated daughter products. Dissolved hydrogen concentrations greater than 1 nmol/L, the concentration above which reductive dechlorination is most favorable, were measured in the landfill wells 6-S-14 and MW-10. Downgradient of the Area 6 landfill, the delineated mildly reducing zone coincides with the area with the area of highest TCA and VC concentrations. Although the downgradient redox conditions are not favorable for continued reductive dechlorination of TCA, the observed TCA concentrations have been well below cleanup standards. The mildly reducing conditions downgradient are favorable for mineralization of VC (Bradley and others, 1998). Aerobic conditions downgradient and to the east of the southern plume (at wells MW-3B and 6-S-3) are also favorable for mineralization of VC.

The changes in redox-sensitive chemicals along a ground-water flow path through the southern plume are typical for sites with active reductive dechlorination followed by mineralization (Weidemeier and others, 1998). The consumption of oxygen, organic carbon, and sulfate and the production of methane, carbon dioxide, and iron (II) within the core of the southern plume confirm that conditions are favorable for the inferred reductive dechlorination (fig. 17).

Table 5. Average 1998 chlorinated volatile organic compound (VOC) concentrations and ratios of primary contaminants to degradation products for selected wells at Area 6, National Air Station Whidbey Island

[Under screen interval column, s is shallow; d is deep, and m, is mixed. Units are all micrograms per liter; **TCA**, 1,1,1-trichloroethane; **DCA**, 1,1-dichloroethane; **DCE**, 1,1-dichloroethane; **TCE**, trichloroethene; ***cis*-DCE**, *cis*-1,2-dichloroethene; **VC**, vinyl chloride; <, less than; >, greater than; –, not analyzed for indicated parameter]

Well identification number	Screen interval	TCA	DCA	DCE	TCA/ DCA	TCA/ DCE	TCE	*cis*-DCE	VC	TCE/ *cis*-DCE	*cis*-DCE/ VC	DCE/ VC
Western plume, shallow and mixed wells, north to south												
6-S-6	s	130	1.6	23	81.3	5.7	330	61.0	<1.0	5.5	>61	>23
6-S-21	m	340	<0.2	17	>1,700	20	0.7	<0.2	<1.0	>3.5	–	>17
6-S-25	s	3,500	5.3	52	660	67	1.8	<0.2	<1.0	>9.0	–	>52
6-S-27	s	16	<0.2	1.7	>80.0	9.4	1.2	<0.2	<1.0	>6.0	–	>1.7
MW-6	s	0.6	<0.2	<0.2	>2.9	>2.9	<0.2	<0.2	<1.0	–	–	–
MW-7	m	380	65	42	5.8	9.0	430	160	<1.0	2.7	>160	>42
N6-38	s	350	1.6	36	220	9.6	31	3.3	<1.0	9.4	>3.3	>36
N6-37	s	48	1.0	2.6	48.0	18	73	5.8	<1.0	13	>5.8	>2.6
PW-1	m	340	8.5	9.1	40	38	210	110	<1.0	1.9	>110	>9.1
PW-3	m	1,100	75	110	15	10	310	100	<1.0	3.1	>100	>110
PW-5	m	3,400	370	200	9.2	17	43	23	<1.0	1.9	>23	>198
Western plume, deep wells, north to south												
6-S-7	d	<0.5	<0.2	<0.2	–	–	<0.2	<0.2	<1.0	–	–	–
6-S-10	d	<0.5	<0.2	<0.2	–	–	<0.2	<0.2	<1.0	–	–	–
6-S-11	d	<0.5	<0.2	<0.2	–	–	<0.2	<0.2	<1.0	–	–	–
6-S-12	d	<0.5	<0.2	<0.2	–	–	<0.2	<0.2	<1.0	–	–	–
6-S-13	d	84	260	9.6	0.3	8.8	1.0	<0.2	<1.0	>5.0	–	>9.6
6-S-22	d	<0.5	<0.2	<0.2	–	–	<0.2	<0.2	<1.0	–	–	–
6-S-26	d	<0.5	<0.2	<0.2	–	–	<0.2	<0.2	<1.0	–	–	–
6-S-28	d	<0.5	<0.2	<0.2	–	–	<0.2	<0.2	<1.0	–	–	–
Southern plume, all wells, north to south												
6-S-3	m	3.6	<0.2	<0.2	>18.0	>18	<0.2	<0.2	<1.0	–	–	–
6-S-14	d	<0.5	1.1	<0.2	<0.5	–	<0.2	<0.2	<1.0	–	–	–
6-S-19	m	8.6	<0.2	<0.2	>43.0	<43	<0.2	<0.2	1.2	–	<0.2	<0.2
6-S-23	s	1.1	6.0	<0.2	0.2	>5.5	<0.2	<0.2	1.1	–	<0.2	<0.2
6-S-29	m	7.8	3.8	<0.2	2.1	>39	<0.2	<0.2	5.1	–	<<0.1	<<0.1
MW-8	m	2.6	2.1	<0.2	1.2	>13	<0.2	<0.2	1.1	–	<0.2	<0.2
MW-9	m	4.0	1.7	<0.2	2.4	>20	<0.2	<0.2	<1.0	–	–	–
MW-10	m	<0.5	21	<0.2	<0.1	–	<0.2	0.3	1.3	<0.7	0.2	<0.2
PW-2	m	3.4	18	<2.0	0.2	>1.7	<2.0	<2.0	1.5	–	<1.3	<0.2
PW-4	m	5.5	4.4	<0.2	1.3	>28	<0.2	<0.2	1.4	–	<0.2	<0.2
PW-6	m	3.0	5.3	0.2	0.6	15	0.3	<0.2	<1.0	>1.5	–	–
PW-7	m	3.7	20	<0.2	0.2	>18.5	0.5	1.4	1.1	0.4	1.3	<0.2
PW-8	m	5.6	1.5	<0.2	3.7	>28	<0.2	<0.2	1.4	–	<0.2	<0.2
PW-9	m	5.6	<0.2	<0.2	>28	>28	<0.2	<0.2	<1.0	–	–	–

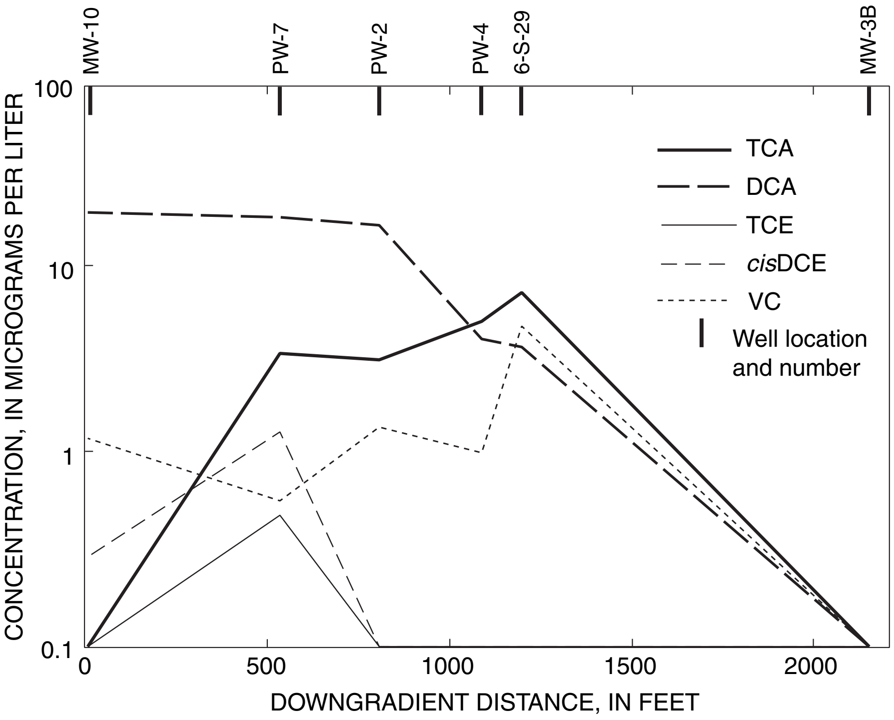

Figure 16. Concentrations of selected chlorinated volatile organic compounds in wells along a flow path through the southern contaminant plume, 1998.

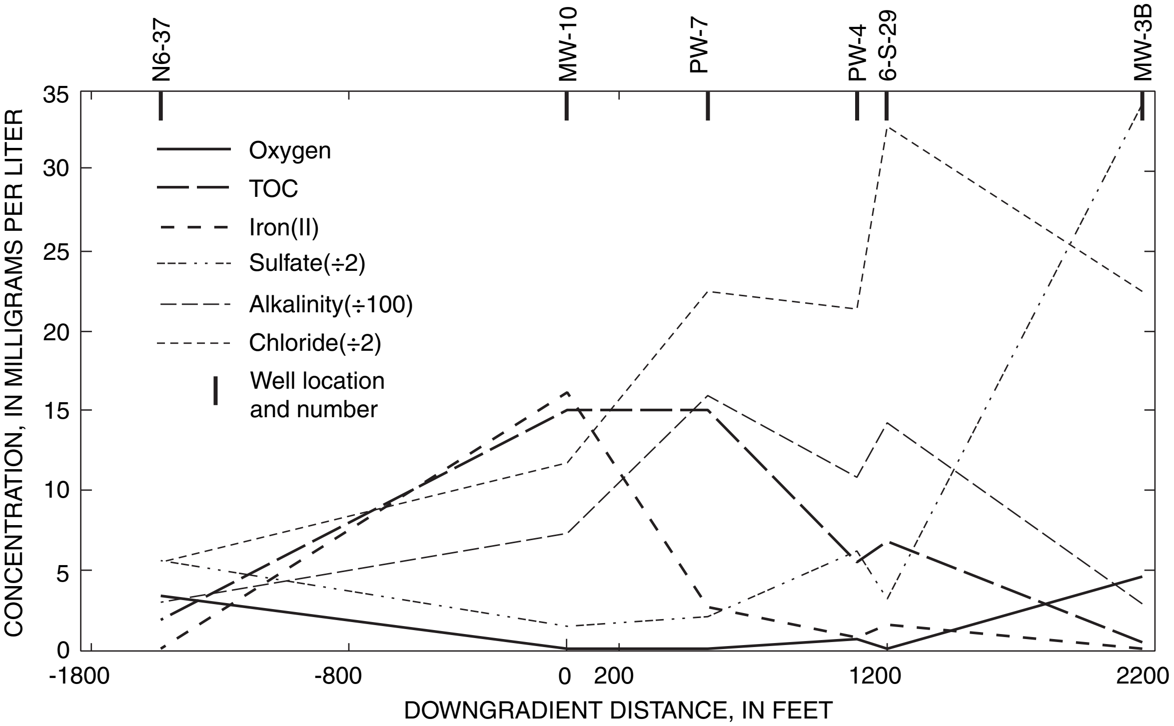

Figure 17. Concentrations of selected redox-sensitive compounds in wells along a flow path through the southern contaminant plume, 1998.

(The flow path illustrated on figure 17 uses redox data from well N6-37 to represent conditions up-gradient from the landfill, although N6-37 lies to the west of the southern contaminant plume flow path—no data were available directly up-gradient of the landfill.) Reduced organic carbon concentrations and replenishment of oxygen from ground-water recharge in the distal part of the southern plume creates the mildly reducing and aerobic conditions favorable for mineralization of VC. Chloride concentrations are highest beneath the Area 6 landfill, which could be interpreted to indicate substantial reductive dechlorination of VOC's in that area. However, the changes in chloride concentrations that would result from degradation of VOC's are likely negligible compared to the observed concentration changes. The observed changes in chloride along the southern plume flow path more likely result from other sources of chloride in the landfill leachate, and from the large amounts of chlorine bleach and sodium hypochlorite that have been used to clean wells at Area 6.

Western Contaminant Plume

It was inferred that natural attenuation was not particularly effective at controlling the migration of VOC's in the western plume, although it may have limited additional plume expansion to the south of the Navy property boundary. Ground-water chemistry evidence confirms that contaminant degradation is not substantial in the shallow part of the western plume, but is more substantial in the deeper part of the leading edge of the plume.

The predominance of parent compounds relative to intermediate degradation products in shallow wells in the western plume (table 5) confirms that reductive dechlorination of TCA and TCE in the upper part of the aquifer has been limited. In clear contrast to the southern plume, TCA and TCE in the western plume were much more abundant than DCA and *cis*DCE in all but one deep well (figs. 18-19). The predominance of parent compounds relative to degradation products is evidence of limited TCA and TCE biodegradation in most of the western plume, although the presence of some DCA and *cis*DCE indicates that limited reductive dechlorination has occurred. The ratios TCA/DCA and TCE/*cis*DCE were consistently higher for shallow wells, compared to the mixed wells (table 5), and the ratios for shallow wells changed little over the length of the plume. The unchanging ratios suggest that either reductive dechlorination is occurring only near the source and the daughter products are recalcitrant, or that reductive dechlorination is occurring along the flow path and the daughter products are degraded at a similar rate. The lower TCA/DCA and TCE/*cis*DCE ratios in mixed wells suggest that reductive dechlorination is more prevalent in the deeper parts of the shallow aquifer, and the predominance of DCA over TCA in the deep well 6-S-13 confirms that conclusion. Chloroethane, the reductive dechlorination by-product of DCA, has been regularly detected only in deep wells 6-S-13 and PW-5, suggesting that some DCA is being further degraded by reductive dechlorination. Few deep wells exist within the core of the western plume up-gradient from 6-S-13, so it is difficult to evaluate at what point contamination reaches the deeper part of the aquifer and how much degradation takes place further north in the plume.

Downgradient of the site boundary, the data for the western plume are too sparse to make definitive conclusions regarding contaminant degradation. Trace amounts of TCA are the only chlorinated VOC detections observed in MW-6, but that lack of contamination cannot necessarily be attributed to degradation. Contaminants from Area 6 may not have been detected at MW-6 because the plume may not have extended that far before implementation of pump and treat, and because the off-base monitoring network may not be extensive enough to observe plume migration. Contaminant concentrations in the shallow well 6-S-27 just south of the site boundary steadily decreased from 140 µg/L in late 1996 to 9.8 µg/L in late 1998, but there is no associated chemistry data to indicate reductive dechlorination in that well. Thus, the decreased concentrations in 6-S-27 were possibly due to pumping at extraction well PW-5. No contaminants were detected during 1998 in the deep well 6-S-28, also possibly due to pumping at extraction well PW-5.

TCE may be more susceptible to reductive dechlorination than TCA or *cis*DCE may be more recalcitrant (resistant to degradation) than DCA, based on the observation that TCE/*cis*DCE ratios were consistently lower than the TCA/DCA ratios in shallow and mixed wells in the western plume. The lack of detectable VC in the western plume suggests that reductive dechlorination of the DCE compounds into VC is negligible, so the probable fate of DCE compounds is direct oxidation with no VC production.

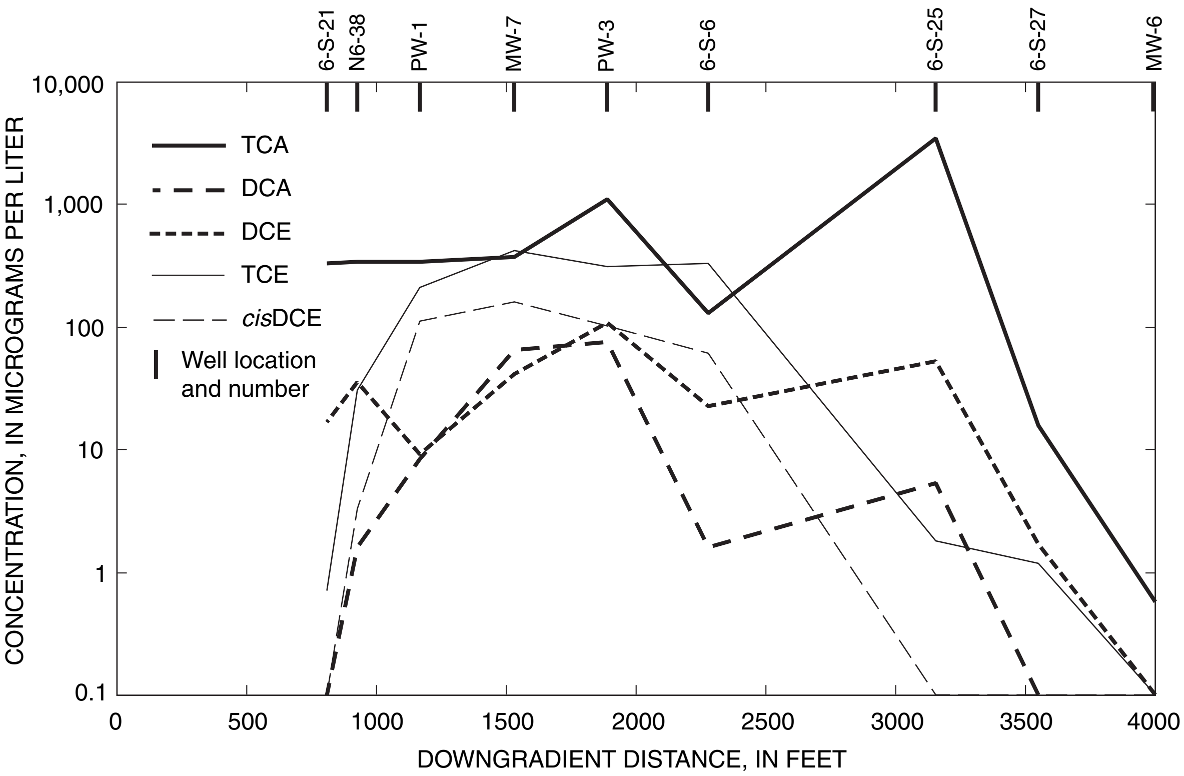

Figure 18. Concentrations of selected chlorinated volatile organic compounds in wells along a shallow flow path through the western contaminant plume, 1998.

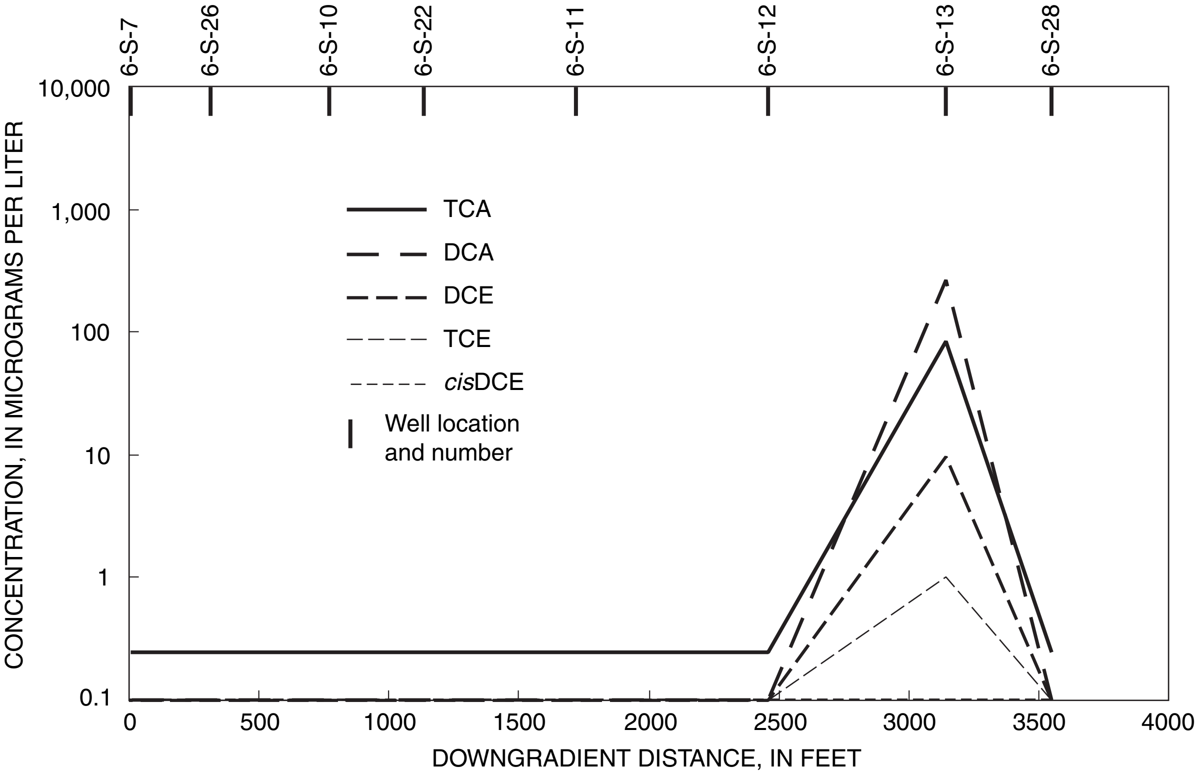

Figure 19 . Concentrations of selected chlorinated volatile organic compounds in wells along a deep flow path through the western contaminant plume, 1998.

Somewhat consistent TCA/DCE ratios in the western plume suggest that the DCE being continually produced by abiotic degradation of TCA is being subsequently mineralized. Concentrations of DCE (the intermediate by-product of abiotic dehydrohalogenation of TCA) in the western plume were 6 to 60 times less than TCA concentrations. There was no obvious spatial pattern in the relative abundance of the two compounds. If it is assumed that the rate of abiotic degradation of TCA is constant and that DCE is recalcitrant, then a steady decrease in the TCA/DCE ratio along a flow path would be observed. Such a pattern was not observed in the western plume; DCE was rarely detected (at concentrations greater than 0.2 µg/L) when TCA concentrations were less than about 10 µg/L. Less can be said about the alternative pathway of abiotic hydrolysis of TCA into acetic acid because acetic acid has not been monitored in the western plume. Also, acetic acid is created and consumed by many other geochemical reactions in ground water, so its presence or absence is not solely diagnostic of TCA degradation.

With regard to redox conditions, the aerobic conditions throughout the upper part of the western plume (fig. 7) are not favorable for reductive dechlorination, which is consistent with the observed low concentrations of reductive dechlorination products. However, redox conditions change from aerobic (at 6-S-27) to anaerobic (at MW-6) downgradient from the Navy property boundary, which is more favorable for reductive dechlorination. Anaerobic conditions favorable for reductive dechlorination (mixed iron and sulfate reduction with H_2 = 0.9 nmol/L) were also identified in the lower part of the plume at the southern site boundary (6-S-13), which is consistent with the greater abundance of DCA over TCA in that well. Similar redox conditions were identified in the next downgradient deep well (6-S-28), so reductive dechlorination could continue in the deeper part of the aquifer south of the site boundary. However, no chlorinated VOC's were observed in 6-S-28 in 1998 due to pump and treat activities, so the potential for reductive dechlorination downgradient of the site boundary is currently unrealized.

The presence of some DCA and DCE in the shallow western plume is possibly a result of past redox conditions in source area. The original source materials were reported as mixed wastes, which likely included nonchlorinated organic carbon compounds such as hydocarbon fuels. The oxidation of that limited carbon source may have resulted in short-lived anaerobic redox conditions beneath the source area which would have been favorable for reductive dechlorination. After waste disposal was discontinued and most of the organic carbon was oxidized, source area ground water returned to ambient aerobic conditions. Also, ground-water samples from mixed wells with long screened intervals are likely a mixture of waters from different depths in the aquifer. Given that redox conditions in the deeper part of the aquifer are more favorable for reductive dechlorination, some of the DCA and *cis*DCE in mixed wells is from ongoing reductive dechlorination at depth.

Changes in concentrations of the redox reactants and products along a flow path in the shallow part of the western plume show limited evidence for reductive dechlorination upgradient of the site boundary (fig. 20), but show potential for reductive dechlorination downgradient of the site boundary. An influx of organic carbon downgradient of the site boundary corresponds with depletion of DO and an increase in iron (II) concentrations from less than 0.01 to 1.32 mg/L (not shown on fig. 20). Sulfate concentrations also increase significantly beneath the Oak Harbor landfill, likely due to gypsum-bearing construction debris buried at the site. The production of carbon dioxide (represented by alkalinity on fig. 20) was limited along the entire shallow ground-water flow path in the western plume, indicating that microbial activity is much less than that occurring in the southern plume. Chloride concentrations generally increased along the flow path, but the increase was likely not due to reductive dechlorination--the changes that would result from reductive dechlorination of VOC's are negligible compared to the observed concentration changes.

Changes in concentrations of redox reactants and products along a flow path in the deep part of the western plume suggest somewhat favorable conditions for reductive dechlorination (fig. 21), but the data are not conclusive because no deep wells are located directly along the ground-water flow path between the source area and the southern base boundary. Data from deep wells alongside the western plume's flow path confirm that redox conditions in the deeper part of the shallow aquifer are a mix of sulfate and iron reduction.

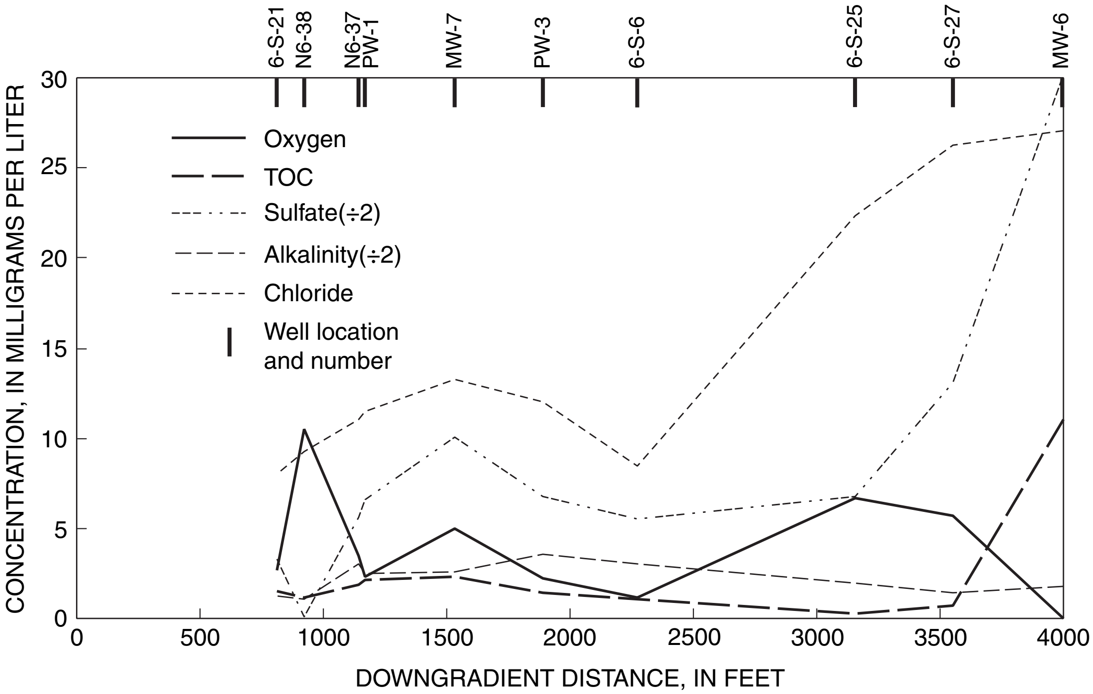

Figure 20. Concentrations of selected redox-sensitive compounds in wells along a shallow flow path through the western contaminant plume, 1998

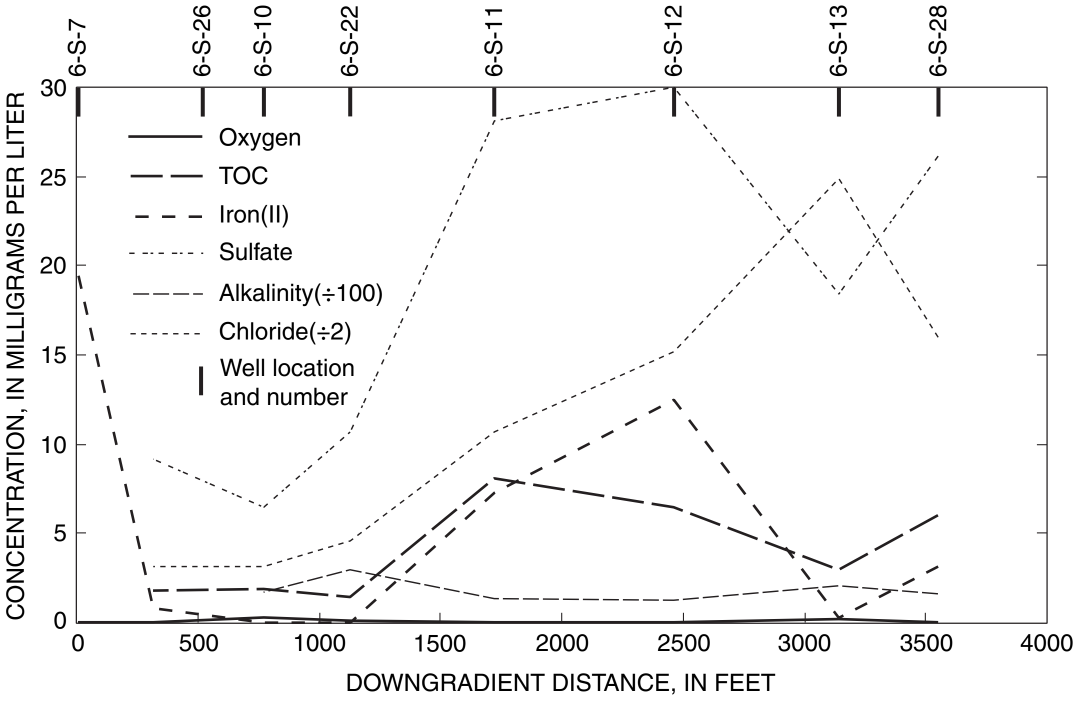

Figure 21. Concentrations of selected redox-sensitive compounds in wells along a deep flow path through the western contaminant plume, 1998.

Ground water up-gradient of the source area had low concentrations of oxygen, sulfate, and organic carbon, but elevated concentrations of iron (II), which is consistent with sulfate reduction near the beginning of the flow path coupled with oxidation of organic carbon from wetland soils. Redox conditions change to predominately iron reduction downgradient from the source area, as indicated by an increase in both sulfate and iron (II) concentrations, and again change back to sulfate reduction at the southern site boundary, as evidenced by decreased sulfate and iron (II). Chlorine concentrations increase steadily along the flow path, but again the expected changes in chloride concentrations from reductive dechlorination of VOC's would be negligible compared to the observed changes.

Limited information about redox reactants and products is available for ground water in the deeper part of the shallow aquifer downgradient of the site boundary, although continued iron- or sulfate-reducing conditions are suggested. Some constituents have been monitored by the City of Oak Harbor at well MW-1, which is screened near the bottom of the aquifer. During 1996–98, TOC concentration ranged from 0.5 to 3.6 mg/L, nitrate was about 0.5 mg/L, manganese ranged from 0.003 to 0.053 mg/L, iron ranged from less than 0.025 to 2 mg/L, chloride was about 40 mg/L, and sulfate was about 80 mg/L. With the exception of greatly increased sulfate concentrations, the chemistry is somewhat similar to that at well 6-S-28, so it would be reasonable to infer that iron- and sulfate-reducing conditions also prevail.

Laboratory Experiments on Biodegradation of Vinyl Chloride and Dichloroethane

The microcosm experiments demonstrated that microorganisms indigenous to the shallow aquifer at Area 6 are capable of significant mineralization of VC under either aerobic or iron-reducing conditions, and are capable of significant mineralization of DCA under either aerobic, manganese-reducing, or iron-reducing conditions. DCA mineralization may be somewhat inhibited under mixed iron/manganese-reducing conditions, but it was still found to be substantial. The VC results and DCA results for aerobic conditions had been demonstrated in previous investigations, but the present study is the first to demonstrate the mineralization of DCA coupled to manganese-reduction.

Vinyl Chloride Mineralization

The results of the VC microcosm experiment showed that microorganisms indigenous to the shallow aquifer at Area 6 were capable of significant mineralization of [1,2-^{14}C] VC to CO_2 under either aerobic or iron-reducing conditions. The recovery of mineralized [1,2-^{14}C] VC as $^{14}CO_2$ is shown on figure 22 and table 6. The differences in recoveries between the active microcosms and the abiotic controls clearly indicate that microorganisms are responsible for the observed VC mineralization. No additional carbon or nutrients were added to the microcosms, so the results demonstrate that the biodegradation can occur under ambient ground-water conditions. The recovery of [1,2-^{14}C] VC as $^{14}CO_2$ in the abiotic controls was, with one exception, between only 3 and 5 percent, indicating little if any abiotic VC mineralization. Microbially mediated mineralization under both aerobic and anaerobic conditions accounted for about 21 percent of the added radio-labeled carbon after 27 days. Microcosm experiments with VC generally do not allow recovery of all added radio-labeled carbon by the end of the experiment, because the majority of the VC (which is a gas at room temperature) tends to volatilize into the headspace above the sediments. Thus, the rates of mineralization are controlled by the slower rate of diffusion of the VC into the sediment-water matrix. The rate of VC mineralization was similar for aerobic and anaerobic conditions at the beginning and end of this experiment. Those results are consistent with previously reported results (Bradley and Chapelle, 1996) for microcosms amended with Fe-EDTA. It is expected that without the iron amendment, the rate of aerobic mineralization would be somewhat greater than that for anaerobic conditions.

Dichloroethane Mineralization

The results of the DCA microcosm experiment showed that microorganisms indigenous to the shallow aquifer at Area 6 were capable of significant mineralization of [1,2-^{14}C] DCA to CO_2 under aerobic, unamended anaerobic, or amended manganese-reducing conditions (fig. 23 and table 7). The DCA mineralization was attributable to biological activity because the final recovery of $^{14}CO_2$ in sterile control microcosms was less than 2 percent. Approximately 92 percent mineralization of [1,2-^{14}C] DCA was observed after 62 days in the aerobic microcosms.

Table 6. Recovery of [1,2-^{14}C] vinyl chloride (VC) as $^{14}CO_2$ in microcosms containing aquifer material from Area 6.

[CO_2, carbon dioxide]

Microcosm	Percentage of $^{14}CO_2$ recovered					
	0 days	**0.25 days**	**0.83 days**	**1.25 days**	**6 days**	**27 days**
Aerobic	6	8	11	11	21	22
Aerobic control	5	4	5	5	9	4
Anaerobic	5	6	10	12	12	23
Anaerobic control	5	3	5	4	5	3

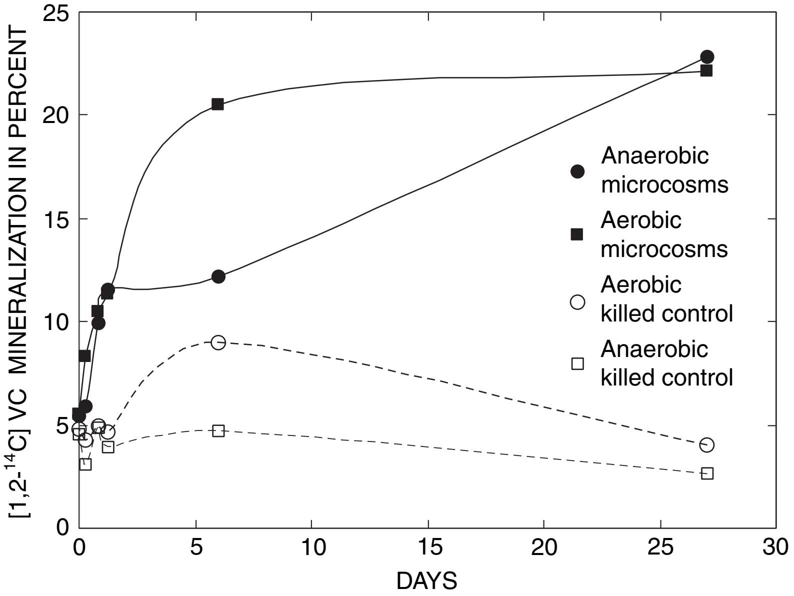

Figure 22. Percentage mineralization of [1,2-^{14}C] vinyl chloride (VC) to $^{14}CO_2$ in microcosms containing aquifer material from Area 6.

Those results are consistent with previously published results. Approximately 65 percent mineralization of [1,2-^{14}C] DCA was observed after 62 days in the unamended anaerobic microcosms. Manganese-reduction was the predominant redox condition under unamended anaerobic conditions, based on the lack of significant DO (less than 3 micromoles (µmol)), nitrate (less than 0.2 µmol), iron (II) (less than 0.2 µmol), sulfate (less than 20 µmol), sulfide (less than 0.2 µmol), and methane (less than 1 µmol per liter of headspace), and significant accumulation of reduced manganese(II) (8 ± 3 nmol produced) in the microcosms. Amendments of MnO_2 stimulated overall microbial respiration and manganese(IV) reduction (as indicated by elevated CO_2 production and a doubling of Mn(II) production; table 7), but they did not significantly affect [1,2-^{14}C] DCA mineralization in this study. Those results indicate that microbial respiration was limited by the bioavailability of Mn(IV) in the aquifer sediments, and they provide additional evidence that manganese reduction was the redox condition in the sediments. However, the amounts of [1,2-^{14}C] DCA mineralized in the anaerobic and anaerobic/amended microcosms did not differ significantly, indicating that Mn(IV) bioavailability did not limit microbial mineralization of [1,2-^{14}C] DCA in these aquifer sediments.

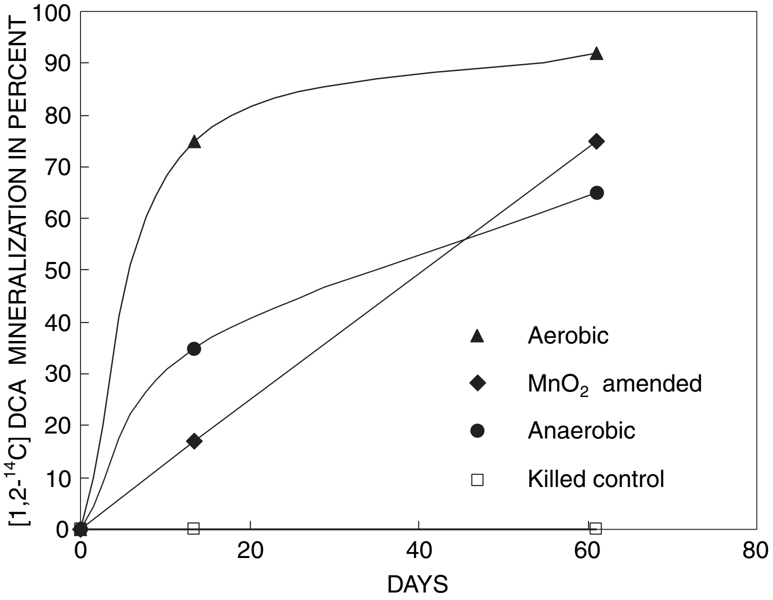

Figure 23. Percentage mineralization of [1,2-^{14}C] dichloroethane (DCA) to $^{14}CO_2$ in microcosms containing aquifer material from Area 6.

No chloroethane, ethane, or ethene were detected in the headspace of the DCA microcosms after 62 days, indicating that DCA is efficiently mineralized to CO_2 without detectable accumulation of volatile intermediates. After 62 days, the amount of DCA remaining in the headspace of all microcosms was measured; no DCA was detected in any experimental treatment, and DCA concentrations did not change significantly over the time period in the sterile controls.

The observed DCA mineralization under anaerobic conditions was not coupled to iron reduction in this study. The possibility of that reaction was examined by amending additional anaerobic microcosms with poorly crystalline iron(III)oxide ($Fe(OH)_3$). The amendments did not result in significant iron(II) accumulation, but did stimulate Mn(II) production above that observed in unamended anaerobic microcosms (table 7). The increased manganese reduction may reflect microbial reduction of iron(III) to iron(II) followed by reoxidation of iron(II) coupled to Mn(IV) reduction. Thus, the observed inhibition of DCA mineralization in $Fe(OH)_3$ amended microcosms is probably due to competitive consumption of MnO_2 by iron(II) oxidation. The fact that the $Fe(OH)_3$ amendment inhibited DCA mineralization also indicates that DCA mineralization was not coupled to iron(II) reduction, and provides additional evidence that the anaerobic mineralization of DCA observed in the present study was coupled to Mn(IV) reduction.

Table 7. Production of dissolved manganese (IV) and the final recovery of [1,2-^{14}C] dichloroethane (DCA) as $^{14}CO_2$ in microcosms containing aquifer material from Area 6

[–, not analyzed at indicated time; CO_2, carbon dioxide; **Mn(II)**, manganese (II)]

	Percent of $^{14}CO_2$ recovered			Moles Mn(II)
Microcosm	**0 days**	**13.5 days**	**6.2 days**	**Produced at 62 days**
Aerobic	0	35	65	8
$Fe(OH)^3$ amended	–	–	44	20
MnO_2 amended	0	17	75	20
Aerobic	0	75	92	0
Control	0	0	0	–

EFFECTIVENESS OF NATURAL ATTENUATION

There is evidence for substantial natural attenuation in the southern contaminant plume and for limited attenuation in the western plume. In this section of the report, a quantitative evaluation of the historical and future effectiveness of natural attenuation is presented. Much of the evaluation relies on application of a simple analytical ground-water flow and transport model, which is described. The historical evaluation included estimates of contaminant travel times and degradation rates in the shallow aquifer without the effects of the pump and treat system, and estimates (where possible) of the mass of contaminants that were removed from ground water by degradation processes alone before 1995. The future evaluation includes estimates of how far downgradient contaminants could migrate if the pump and treat activities were stopped, and a discussion of the effectiveness of natural attenuation for meeting existing site-specific remediation objectives. One critical criterion for the effectiveness of natural attenuation is that it must reduce the potential risk to existing and future ground-water users (potential receptors) located downgradient of the site. At Area 6, the nearest known existing down-gradient water-supply wells in the shallow aquifer are located 1,800 ft from the site boundary in the vicinity of the southern plume, and 2,000 ft from the site boundary in the vicinity of the western plume. Washington State regulations prohibit new wells within 1,000 ft of a solid waste landfill [WAC 173-160-171, (3)(b)(vi)] and the nearest existing wells are just within 1,000 ft of the former Oak Harbor landfill, so no future downgradient wells will be closer than existing wells. Therefore, the nearest potential receptors downgradient of the site boundary near the southern plume are 1,800 ft away, and the nearest potential receptors downgradient of the site boundary near the western plume are 2,000 ft away. Those are the maximum distances available for natural attenuation processes to reduce ground-water contaminant concentrations to below levels of concern.

Ground-water Flow and Transport Model

The analytical model for flow and transport by Domenico (1987) was used to explore various contaminant migration and degradation issues at Area 6 for non-pump and treat conditions. Although less sophisticated than a three-dimensional numerical model, the Domenico model is a reasonable tool to simulate the relatively uniform ground-water flow conditions at Area 6 that occur when the Navy's extraction wells are not operating. The model simulates advective contaminant transport, three-dimensional dispersion, adsorption, and a first-order approximation for contaminant degradation. The Domenico model was run within the BIOSCREEN (Version 1.4) ground-water flow and transport computer program (U.S. Environmental Protection Agency, 1998). In addition to the features of the Domenico model, the BIOSCREEN program simulates either a constant contaminant concentration at the source or a declining contaminant mass and concentration at the source as approximated by a first-order decay expression. Required model input includes parameters related to source concentration (and mass if a declining source is simulated), hydrogeology, dispersion, and adsorption.

Although the BIOSCREEN program has most often been used to evaluate natural attenuation of petroleum hydrocarbons, the Domenico analytical model embedded within BIOSCREEN is applicable for simulating transport of any conservative contaminant, or any decaying contaminant that can be approximated by a first-order expression. The BIOSCREEN program also includes an instantaneous reaction model that can be superimposed on the Domenico model to specifically simulate degradation of petroleum hydrocarbons, but that, and other features programmed specifically designed for the simulation of petroleum hydrocarbon transport, were not used for this investigation.

The following parameter values were used for contaminant migration and degradation simulations at Area 6. An average horizontal hydraulic gradient of 0.002 ft/ft was used for all simulations. That value was estimated from observed prepumping water-level data from January 1995, and is appropriate for the nonpumping simulations completed for this evaluation. A horizontal hydraulic conductivity of 87 ft/d was used, which is the average value from 1998 aquifer tests (Foster Wheeler Environmental, 1998b). Selected simulations were also run using minimum and maximum hydraulic conductivity estimates of 47 and 126 ft/d, which also came from 1998 aquifer tests. A porosity of 0.25 was assumed based on the medium sand texture of the shallow aquifer materials. A longitudinal dispersivity of 24.5 ft was estimated by assuming a plume length of 1,000 ft in the Xu and Eckstein (1995) formula. Transverse and vertical dispersivities were assumed equal to 10 percent and 1 percent of the longitudinal dispersivity, respectively. Retardation factors ranged from 1.3 to 1.8, depending on the contaminant being simulated. Those factors were calculated from organic carbon fractions of aquifer materials of 0.001 to 0.0003 (URS Consultants, 1993b), estimated porosity and bulk density values of 0.25 and 1.9 kg/L, and reported K_{oc} values of 3 to 57 L/kg (Mackay and others, 1992). The source area contaminant concentrations and masses varied for the particular scenarios simulated, as described in the following sections.

Simulated Contaminant Migration and Degradation in the Southern Plume

Historical Vinyl Chloride Migration and Degradation in the Southern Plume

Simulations with the analytical model showed that enough time had passed before implementation of pump and treat to allow migration of VC to the downgradient wells MW-3B and AHI, so contaminant degradation is a plausible explanation for the lack of VC detections in those wells. VC migration in the southern plume was simulated beginning with a source area located in the shallow aquifer at the southern margin of the landfill. The cross-sectional dimensions of the source area were assumed to be 750 ft wide and 20 ft thick. Ground water flowing within the middle 250 ft of the source area was assigned a constant VC concentration of 54 µg/L, which corresponds to the highest historical concentration observed. Ground water flowing from the remainder of the source area was assigned a constant VC concentration of 5.4 µg/L. No degradation of VC in the plume was simulated. Results using average hydraulic conductivity estimates indicated that the leading edge (1 µg/L) of the plume would extend 1,100 ft downgradient to the AHI wells in 4 years, and would extend 2,000 ft downgradient to well MW3-B in 8 years. Using minimum and maximum hydraulic conductivity estimates, the travel-time to the AHI wells would range from 2.5–7 years, and the travel time to well MW-3B would range from 5.5–14 years. The Area 6 landfill opened in 1969, and it likely took a few years before VC appeared in ground water. Assuming that VC was not present in ground water until as late as 1980, it should have been detectable in the AHI wells around 1983–88, and it should have been detectable in well MW3-B around 1986–96. Given that VC has never been detected in those downgradient wells, it is likely that VC was degraded to nondetectable concentrations. Thus, contaminant degradation is a plausible explanation for the lack of VC detections in wells MW-3B and the AHI, although it is still possible that VC could have migrated beneath or around the downgradient wells.

A sensitivity analysis of selected Domenico model parameters showed that simulated travel times are most sensitive to changes in hydraulic conductivity estimates (fig. 24). The model parameters evaluated included hydraulic conductivity, effective porosity, dispersivity, and retardation. For the sensitivity analysis, hydraulic conductivity was varied between 47 and 126 ft/d according to the variation found in aquifer test results reported by Foster Wheeler Environmental (1998b). Effective porosity was varied between 0.16 and 0.46 according to the range for medium sands reported by McWhorter and Sunada (1977). Longitudinal dispersivity was increased and decreased by a factor of two from the original estimate of 24.5 ft, and the ratios of lateral and vertical dispersivity to longitudinal dispersivity where held constant. Retardation factors were varied between 1 (the minimum possible value) and 2.25 (25 percent greater than the largest original estimate for the most sorptive contaminant of concern, TCA). The variation in simulated VC travel times between the southern margin of the Navy landfill and the AHI wells that resulted from changes in hydraulic conductivity was greater than the variation resulting from changes in the other parameters (fig. 24). Thus, rather than describing the sensitivity of all subsequent simulation results to all model parameters, only the sensitivity to hydraulic conductivity is discussed. It is recognized that simulation results could vary by an even greater amount if more than one parameter was changed to be most favorable for either very fast or very slow contaminant migration. However, the likelihood of multiple parameter values being biased in a consistent way decreases as more parameters are considered.

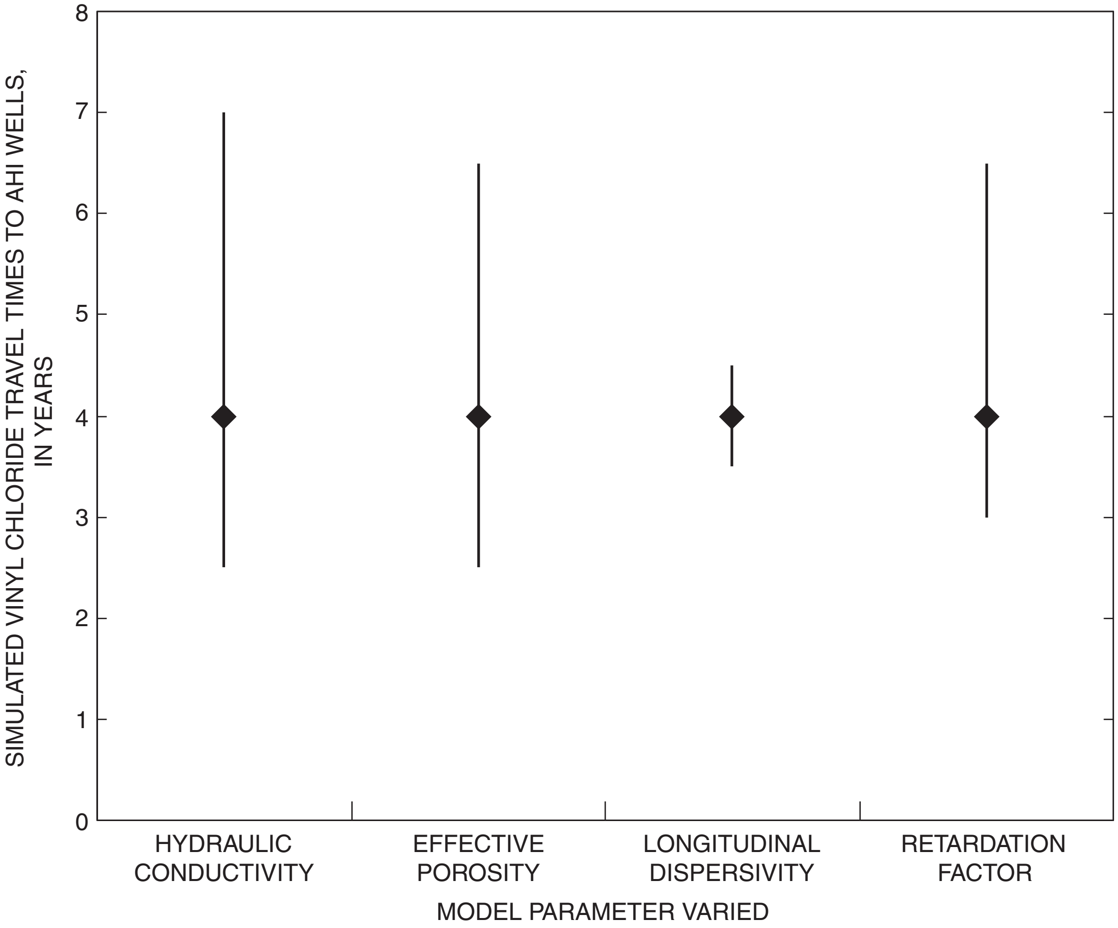

Figure 24. Variation in simulated vinyl chloride (VC) travel times resulting from changes in selected Domenico model parameter values.

Travel times are for the 1,100 feet between the southern margin of the Navy landfill and the AHI wells. The simulated travel time resulting from the original parameter values was 4 years.

Future Vinyl Chloride Migration and Degradation in the Southern Plume without Pump and Treat

Simulations with the analytical model indicated that natural attenuation will likely keep VC in the southern plume from being transported downgradient to potential receptors if pump and treat were discontinued. Future VC migration in the southern plume was simulated beginning with a source area representing observed 1998 conditions at the southern site boundary. The cross-sectional dimensions of the source area were assumed to be 1,250 ft wide and 20 ft thick. Ground water flowing within the middle 250 ft of the source area was assigned a constant VC concentration of 10 µg/L, and ground water flowing from the remainder of the source area was assigned a constant VC concentration of 1 µg/L. Degradation of VC in the plume was simulated because of the abundance of field and laboratory evidence that suggest that VC can be rapidly mineralized in Area 6 sediments under the ambient aerobic and iron reducing conditions. However, a quantitative estimate of historical VC field degradation rates could not be made directly from the existing data at Area 6. The two most commonly used methods to estimate degradation rates from field data—the conservative-tracer approach (Wiedemeier and others, 1996) and the dispersion-model approach (Chapelle and others, 1996)—could not be applied at Area 6. A suitable conservative tracer was not found in the Area 6 data to use the first method, and the observed VC data were too widespread and too variable over time to use the second method.

Because a field degradation rate could not be estimated from historical data, the solute-transport model was instead used to determine the slowest degradation rates that would be required to keep steady-state VC concentrations below levels of concern before reaching potential receptors 1,800 ft downgradient from the site boundary. The VC concentrations of concern were 0.02, 0.1, and 1 µg/L, which are the specified site cleanup level, the compliance cleanup level, and the EPA Method 601 detection limit, respectively (URS Consultants, 1993a). Simulation results indicated that the VC degradation half-lives needed to achieve the various cleanup levels were 1.1, 1.5 and 5 years, corresponding to first-order degradation rates of 0.63, 0.46, and 0.14 per year. The rate needed to achieve the 0.02 µg/L site clean-up level is comparable to the average field attenuation rate constant for VC of 0.6 per year compiled by Wiedemeier and others (1988). The rate needed to achieve the 1 µg/L level method detection limit is a slower rate than all VC degradation rates reported by the same reference. Thus, despite the uncertainty in the absolute value of the VC degradation rate in the southern plume, the evaluation indicated that it is highly probable that natural attenuation will keep VC concentrations below levels of concern before reaching potential receptors if pump and treat were discontinued.

Future 1,1,1-Trichloroethane Migration and Degradation in the Southern Plume without Pump and Treat

TCA has been the only other consistently detected contaminant of concern in the southern plume. TCA concentrations have regularly been less than 10 µg/L, which is well below the site cleanup level of 200 µg/L. However, one of the abiotic degradation products of TCA is DCE. Because the site cleanup level for DCE is only 0.07 µg/L, downgradient DCE concentrations may be of concern. The kinetics of DCE formation and degradation are not well known at Area 6, but observed TCA and DCE concentrations (table 5) show that DCE concentrations have consistently been between 2 to 18 percent of TCA concentrations throughout the site. The ratio of DCE to TCA showed no apparent relation to redox conditions, and the ratio did not change consistently along flow paths or over time. Those observations suggest that the rate of TCA degradation into DCE is about the same as the rate of DCE mineralization. Thus, as long as TCA concentrations are below 0.4 to 3.5 µg/L at the location of potential receptors, DCE concentrations will likely be less than 0.07 µg/L before reaching potential receptors.

Simulations of TCA migration in the southern plume with the analytical model indicated that natural attenuation likely will reduce the 10 µg/L of TCA currently observed at the site boundary to about 1.4 µg/L by the time the plume reaches potential receptors. (The TCA sorption and degradation parameters used for this simulation are described in detail in the subsequent section of this report). Thus, DCE concentrations could possibly exceed the 0.07 µg/L clean-up standard at potential receptor locations if pump and treat were discontinued, although the concentrations would remain well below the method detection limit (1 µg/L) and the USEPA drinking water MCL (7 µg/L).

Simulated Contaminant Migration and Degradation in the Western Plume

Historical 1,1,1-Trichloroethane Migration and Degradation in the Western Plume

Simulations of TCA migration in the western plume indicated that the lack of TCA before 1995 at the furthest downgradient well MW-6 is not clear evidence for contaminant degradation beneath the Oak Harbor landfill—the data can be explained by assuming a reasonable flow scenario with conservative transport of TCA. That conclusion was based on simulated pre-pumping travel times of TCA in the western plume between the source area (well N6-38), the southwestern site boundary (well 6-S-25), and the downgradient margin of the Oak Harbor landfill (well MW-6). The simulations began with a saturated zone source area located at N6-38 that was 500 ft wide and 20 ft thick. TCA concentrations at the source were set to be constant at 1,000 µg/L in the center 250 ft, and ranged down to 10 µg/L at the margins. No degradation of TCA was simulated. Results showed the average TCA travel time from the source area to the site boundary was about 12 years (estimates ranged from 8 to 20 years using the maximum and minimum hydraulic conductivity estimates). Observed data show that the leading edge of the TCA plume reached the southwestern site boundary in about 1991, so it can be concluded that contaminants were deposited in source-area ground water around 1979 (dates range from 1971 to 1983 using the maximum and minimum hydraulic conductivity estimates). The simulated travel times are in agreement with the reported disposal of contaminants on the surface in the early 1970's through the early 1980's. Simulations further showed that the estimated arrival time of TCA at the downgradient well MW-6 (at greater than 1 µg/L) was 1995, which corresponds to when pump and treat was implemented. Although the capture zone of the Navy extraction wells did not extend to MW-6, the extraction wells did quickly alter the ground-water flow field (Foster Wheeler Environmental, 1998a) so that contaminants south of the site boundary may have been drawn eastward of MW-6.

Additional simulations of TCA migration in the western plume suggest that degradation processes significantly reduced TCA concentrations before pump and treat was implemented in 1995. The presence of TCA degradation products in the western plume is direct evidence that both reductive dechlorination and abiotic degradation have occurred. However, the amount of degradation was difficult to quantify directly because there are few data regarding the original contaminant mass or early plume development, and because the 1989–95 contaminant data lack precision. (For example, field duplicate samples in 1993 showed from 1,900 to 5,300 µg/L TCA in well 6-S-6, and showed from 840 to 1,500 µg/L *cis*DCE in well PW-1.) Thus, the analytical model was used to quantify contaminant degradation.

To simulate TCA migration and degradation in the western plume, it was necessary to first estimate the probable decline in source area TCA concentrations that resulted from cessation of waste disposal. Source area TCA concentrations at well N6-38 between 1989 and 1993 varied between 32,000 and 1,500 µg/L. The overall trends in TCA concentrations from 1989–93 and from 1989–98 were downward. The 1989–98 TCA data from well N6-38 fit a first-order exponential approximation (R^2 = 0.78, half-life = 1.5 years), although the highly variable 1989-93 data are not nearly as well represented by the same expression (R^2 = 0.08, half-life = 4 years). Given that some of the concentration decline after 1995 could be attributed to the pump and treat system, it was assumed that the source area TCA concentrations declined according to a first-order exponential decay equation with a half-life of around 2 to 4 years. The conceptual model employed is that source area concentrations declined primarily due to dissipation after cessation of waste disposal, rather than due to TCA degradation.

TCA migration in the western plume before 1995 was then explored by using the assumption of a declining source in the analytical model. The BIOSCREEN program allows simulation of a declining source with a first-order exponential equation, but estimates of the original TCA mass and dissolved concentration in the source area are required. A series of simulations were completed using different estimates of (1) source half-life, (2) original (1980) TCA mass and concentrations in the source area, and (3) TCA degradation half-life in the plume. The goal of the simulations was to match the observed 1995 TCA data along the centerline of the plume (wells PW-1, MW-7, PW-3, 6-S-6, and 6-S-25). The resulting best-fit simulation results compared well to the observed 1995 TCA concentrations (fig. 25), as well as to the 1991-93 source area data from N6-38 (results not shown). The estimated model parameters required to get those results were all reasonable; the source half-life was 4 years, the original TCA mass was 2,000 kilograms (kg), the original source area TCA concentration was 32,000 µg/L, and the TCA degradation half-life in the plume was 5 years.

Results from an additional simulation using identical source area parameters but assuming no TCA degradation in the plume indicated that degradation processes must have substantially reduced TCA concentrations throughout the plume before pump and treat was implemented in 1995. Simulated TCA concentrations throughout the plume were greatly overestimated by the model (fig. 25) when no degradation was assumed. When degradation was included into the model with the calibrated 5-year TCA half-life (fig. 25), observed TCA concentrations were well simulated. Simulation results indicated that about 1,500 kg of TCA was degraded in ground water over the 15-year period 1980–94.

Finally, an alternative conceptual model that assumed a constant TCA source over time and no TCA degradation within the western plume was tested with the analytical model and discarded. When the transport model was run with an assumed continuous source area TCA concentration equal to the average 1989-93 concentration of 14,000 µg/L, the relatively steady 2,000–4,000 µg/L TCA concentrations observed in 1995 throughout the length of the plume were oversimulated by about a factor of two. When the continuous source area TCA concentration was decreased to 4,000 µg/L, the observed 1995 concentrations were well simulated, but acceptance of those results requires rejecting all reported TCA source-area concentrations greater than 4,000 µg/L.

In summary, the historical TCA simulations in the western plume indicated that degradation processes substantially reduced TCA concentrations throughout the western plume before pump and treat was implemented, and that the source area concentrations have declined exponentially since the cessation of waste disposal. A declining source mass and concentration is consistent with the known cessation of waste disposal and the observed data, and the declining source assumption is consistent with the previously reported TCA concentrations of greater than 4,000 µg/L in the source area.

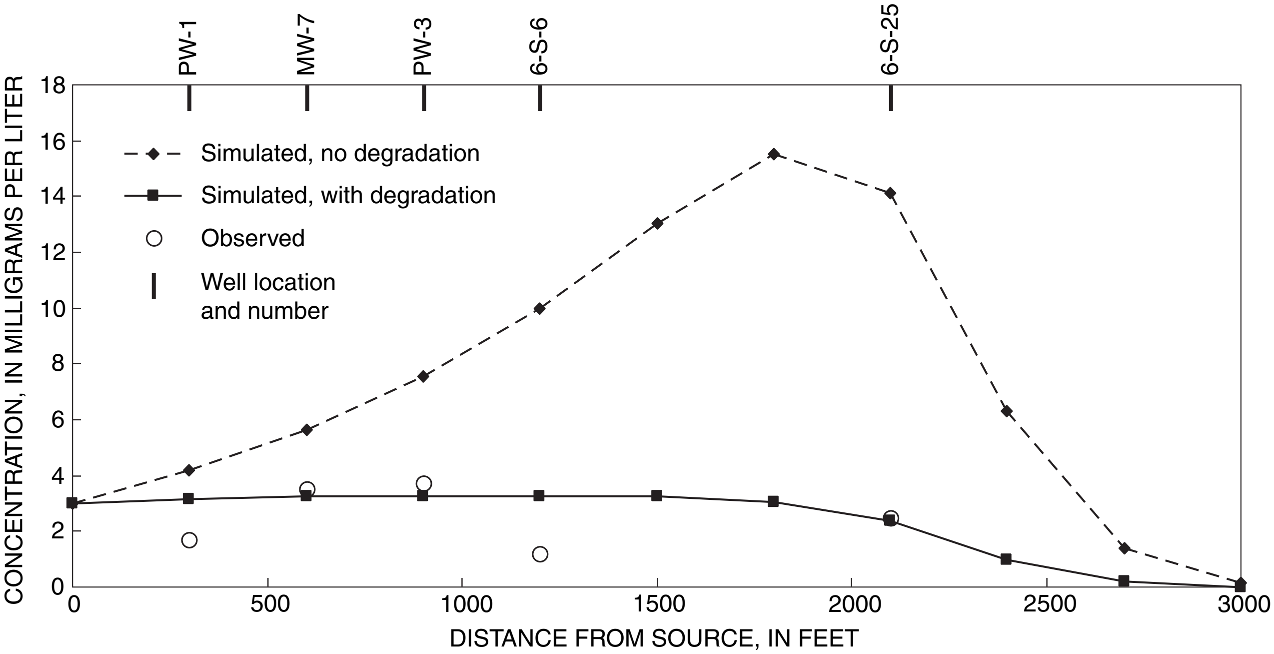

Figure 25. Simulated 1,1,1-trichloroethane (TCA) concentrations with and without degradation, and observed 1995 concentrations of TCA along the centerline of the western contaminant plume.

A decaying source was assumed.

Future 1,1,1-Trichloroethane Migration and Degradation in the Western Plume without Pump and Treat

Although the historical simulations indicated substantial TCA degradation in the western plume, contaminant chemistry data indicate that natural attenuation processes have not been effective enough at controlling TCA migration in the western plume. Thus, the flow and transport model for the western plume was used to estimate a target TCA concentration that pump and treat needs to attain before natural attenuation could become an effective alternative. Multiple model runs were made (with no pump and treat simulated) to determine what the TCA concentration needs to be at the downgradient site boundary in order for natural attenuation to reduce that target concentration to below the 200 µg/L cleanup level when the plume reaches potential ground-water receptors. The closest potential receptors are located about 2,000 ft downgradient from the southwest corner of Area 6, as described previously in this report. The critical model parameter for this exercise was the TCA degradation rate, and (as was the case for the southern plume) the two commonly used methods to estimate degradation rates from field data could not be applied for the western plume. A suitable conservative tracer was not found in the data to use the first method, and the plume did not likely reach steady-state before the pump and treat began, which precludes using the second method. However, the simulation results from preceding model runs yielded a degradation half-life estimate of 5 years, which is comparable to the theoretical TCA abiotic degradation rate of 1 to 5 years. Thus, the conceptual model employed in the future TCA simulations assumed that abiotic degradation with a contaminant half-life of 5 years is the primary TCA degradation pathway, and that the abiotic degradation is independent of redox conditions in the plume.

The future simulations indicate that TCA concentrations in the western plume at the southern site boundary need to be lowered from the average 1998 value of 3,500 µg/L down to around 1,000 µg/L in order to be attenuated to 200 µg/L within 2,000 ft. The transport simulations assumed a constant 1,000 µg/L source and a 5-year TCA degradation half-life. If the maximum hydraulic conductivity estimate of 126 ft/d were assumed, then the TCA concentrations at the site boundary would need to be lowered to 650 µg/L. The site cleanup level of 800 µg/L for DCA is relatively high and was already met throughout Area 6 in 1998, so future DCA concentrations are not likely to be a concern. However, the site cleanup level of 0.07 µg/L for DCE is very low, so the future DCE concentrations will be of concern even if TCA concentrations are attenuated to below 200 µg/L at the downgradient receptors.

TCA concentrations in the western plume at the southern site boundary would need to be reduced to between 3 to 25 µg/L in order for DCE not to exceed 0.07 µg/L at the location of potential downgradient receptors. As previously explained, the empirical relation between TCA and DCE suggests that DCE concentrations are consistently maintained at between 2 to 18 percent of TCA concentrations throughout the site. Thus, in order for DCE concentrations to be below 0.07 µg/L, TCA concentrations must be below 0.4 to 3.5 µg/L. Assuming the same TCA migration and degradation conditions described in the previous paragraph, the analytical model was used to determine that TCA concentrations at the southern site boundary need to be between 3 to 25 µg/L in order for TCA not to exceed 0.4 to 3.5 µg/L within 2,000 ft downgradient of the site boundary. Natural attenuation may be effective at preventing downgradient receptors from exposure to excess TCA concentrations if on-base concentrations are reduced to below 1,000 µg/L, but the on-base TCA concentrations must be reduced to between 3 and 25 µg/L to prevent unacceptable downgradient exposure to DCE.

Historical Trichloroethene Migration and Degradation in the Western Plume

In contrast to TCA migration, little TCE had migrated south of the site boundary in the western plume before the onset of pump and treat in 1995. Observed data and modeling results indicate that TCE reached ground water beneath Area 6 about 5 years after TCA did. The data show that the leading edge of the TCE plume had migrated only half the distance to the southern site boundary (well 6-S-6) by 1990, and did not extend to the site boundary (well 6-S-25) until 1995. Simulations with the analytical model indicate that the TCE travel time between the source area (well N6-37) and well 6-S-6 is about 5 years, and the estimated travel time between the source area and well 6-S-25 is about 10 years. Working backwards from the known arrival times, it was estimated that TCE reached source-area ground water in about 1985. That date corresponds to the reported storage of drummed wastes in the vicinity of N6-37 (active through 1986), and to the latest dates of reported liquid waste disposal (active through the early 1980's).

As was found for TCA, simulations of TCE migration in the western plume indicate that degradation processes substantially reduced TCE concentrations before pump and treat was implemented in 1995. The presence of the degradation product *cis*DCE in the western plume is direct evidence that some reductive dechlorination of TCE had occurred, so the analytical model was used to quantify the TCE degradation. Because the observed source area concentrations of TCE consistently decreased between 1989–95, the conceptual model of a fixed source concentration was deemed inappropriate and a declining source term was used in the simulations. The reported source area concentrations of TCE (at N6-37) decreased from 1,800 µg/L in 1989 to 450 µg/L in 1995, and the 1989–98 data fit a first-order exponential approximation well ($R^2 = 0.91$, source half-life = 1 year), as did the 1989–95 data ($R^2 = 0.79$, source half-life = 2 years). Thus, TCE migration in the western plume before 1995 was simulated by using the assumption of a declining source in the analytical model. A series of simulations was completed using different estimates of original (1985) TCE mass and concentrations in the source area, and different values for TCE degradation half-life in the plume. The goal of the simulations was to match the observed 1995 TCE data along the centerline of the western plume. The best-fit simulation results compared well to the observed 1995 TCE concentrations (fig. 26), and the estimated model parameters required to achieve those results were all reasonable; the original TCE mass was 300 kg, the original source-area TCE concentration was 10,000 µg/L, and the TCE degradation half-life was 5 years. As was found for TCA simulations in the western plume, the results from a simulation that assumed no TCE degradation in the plume greatly over-simulated TCE concentrations throughout most of the plume (fig. 26). When degradation was simulated with the 5-year TCE half-life, results showed that about 220 kg of TCE dissolved in ground water was degraded over the 10-year simulation period 1985–94.

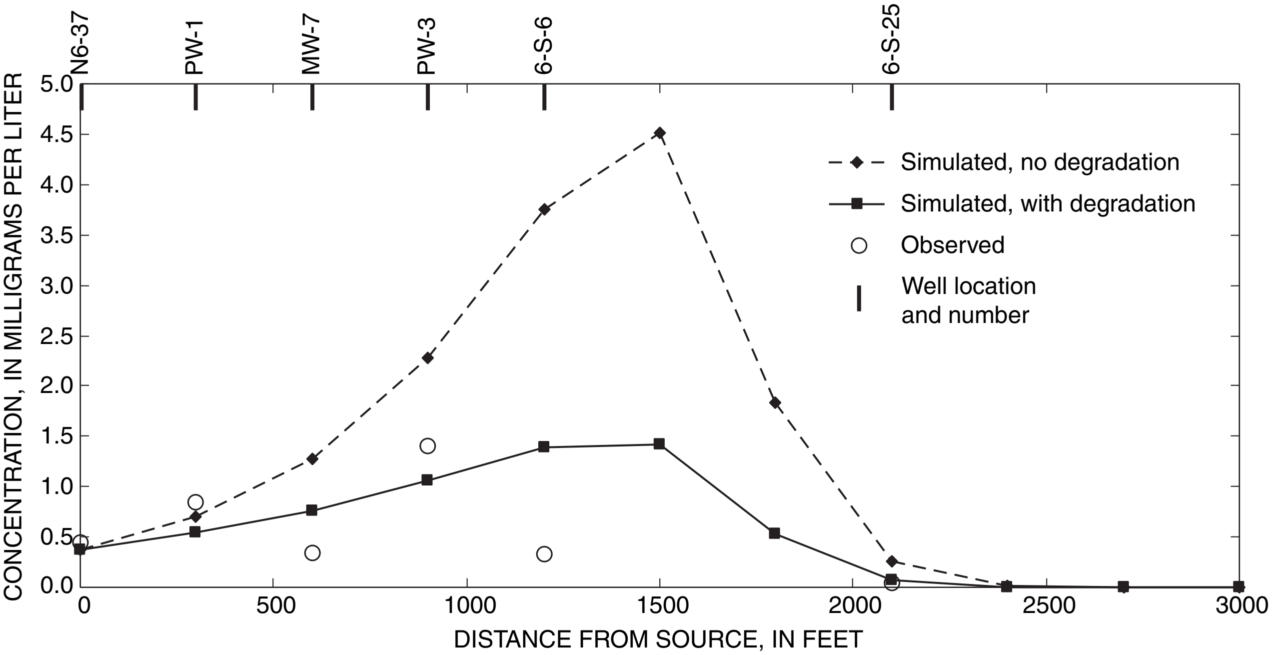

Figure 26. Simulated trichloroethene (TCE) concentrations with and without degradation, and observed 1995 concentrations of TCE along the centerline of the western contaminant plume.

A decaying source was assumed.

Future Trichloroethene Migration and Degradation in the Western Plume without Pump and Treat

The flow and transport model for the western plume was then used to estimate a target TCE concentration that pump and treat must attain before natural attenuation could become an effective alternative. Multiple model runs were made (with no pump and treat simulated) to determine what TCE concentrations must be at the southern site boundary in order for natural attenuation to reduce the concentrations to below the 5 µg/L clean up level when the plume reaches potential ground-water receptors. The most nearby potential receptors are located about 2,000 ft downgradient from the southwest corner of Area 6. The critical model parameter for this exercise was the TCE degradation rate, and the 5-year half-life resulting from the simulation of prepumping TCE migration was assumed. Unlike the conceptual model for TCA degradation, the future scenarios for TCE degradation do rely on reductive dechlorination as the primary degradation pathway, so the degradation is dependent on redox conditions in the plume. As was presented earlier, redox conditions in the western plume downgradient of the site boundary appear to be favorable for reductive dechlorination of TCE, but data from that area are limited.

Simulation results indicate that TCE concentrations at the site boundary must be around 40 µg/L in order to be attenuated to below 5 µg/L when the plume reaches potential ground-water receptors. The average 1998 TCE concentration at the site boundary was 43 µg/L, but the highest concentrations are currently located up-gradient from the property line. If the maximum hydraulic conductivity estimate of 126 ft/d was assumed, then the TCA concentrations at the site boundary would need to be around 20 µg/L. The site cleanup level of 70 µg/L for the TCE reductive dechlorination product *cis*DCE is relatively high and would likely be met throughout Area 6 if TCE concentrations were 40 µg/L or less. Because VC has only rarely been detected in the western plume, and because the strong reducing conditions required to produce VC have not been observed in the western plume, the most probable fate of *cis*DCE is direct mineralization to carbon dioxide, water, and chloride. Thus, unlike TCA, effective natural attenuation of TCE alone will likely result in effective natural attenuation of its degradation products also.

Effectiveness at Meeting Area 6 Remediation Objectives

Natural attenuation is considered effective when natural processes can meet site-specific remediation objectives in a time period that is reasonable compared to other alternatives (Weidemeier and others, 1998). There are four relevant site-specific remediation goals for VOC's in ground-water specified in the 1993 ROD for Area 6 (URS Consultants, 1993a):

- Reduce concentrations of contaminants (TCE, 1,1,1-TCA, 1,1-DCA, 1,1-DCE, *cis*-1,2-DCE, and VC) that have already migrated into the shallow aquifer with the ultimate goal of meeting state and federal drinking water standards at point of compliance locations
- Prevent the further spread of VOC's in the shallow aquifer and treat extracted water to meet state and federal standards prior to discharge
- Reduce the potential risk to existing and future ground water users located downgradient of the site
- Prevent further migration of contaminated ground water across the site boundary and into the lower aquifers

There are six relevant ground-water cleanup levels established:

Contaminant		Cleanup level (µg/L)
Trichloroethene	(TCE)	5
1,1,1-trichloroethane	(TCA)	200
1,1-dichloroethane	(DCA)	800
1,1-dichloroethene	(DCE)	0.07
cis-1,2-dichloroethene	(*cis*DCE)	70
Vinyl chloride	(VC)	[1]0.02

[1]Compliance cleanup level is 0.1.

The conditional points of compliance where the cleanup levels are to be met are approximately at the perimeter of the former Navy landfill operations area, and at the perimeter of a 500-ft-diameter circle centered halfway between wells N6-37 and N6-38 and not to exceed the western property boundary. The time period required for the pump and treat remediation system to meet remediation goals was not specified in the ROD because of uncertainties associated with the site.

The projected effectiveness of natural attenuation alone at meeting the Area 6 remediation objectives is described in this section of the report. For comparison, the current and projected effectiveness of the landfill cap and the pump and treat system at meeting those same objectives is also discussed.

Southern Contaminant Plume

Natural attenuation in the southern contaminant plume could likely achieve all remediation objectives as effectively as pump and treat could, with the exception that the cessation of pump and treat would allow some off-base migration of VC above the cleanup level. However, this small amount of VC would likely be mineralized before reaching potential receptors. In the southern plume, contaminant mass and concentrations have been reduced substantially by natural attenuation processes, and similar reductions will likely continue into the future with or without pump and treat. Concentrations of VC were relatively low before the onset of active remediation, and there has been no additional reduction that could be attributed to pump and treat. Prior to the onset of pump and treat, VC concentrations (up to 4.5 µg/L) exceeded cleanup levels at four on-base wells outside of the compliance zone, and concentrations had changed little through 1998. Concentrations of DCE sporadically exceeded cleanup standards at a few locations in the southern plume between 1995 and 1998, but the highest concentration was 1.6 µg/L.

It is not known how long it will take for either natural attenuation or pump and treat to reduce VC concentrations to below cleanup standards throughout the entire southern plume area. Although the landfill cap has not had an observable effect on reducing contaminant concentrations in the southern plume, it could be presumed that the cap will eventually prevent migration of contaminants from the landfill to ground water. The 1991–98 contaminant data suggest that the

VC plume has been at steady-state for many years; VC production by reductive dechlorination of source materials has been balanced by VC mineralization within the extent of the existing plume. Once the landfill cap successfully limits the supply of contaminants to the southern plume, mineralization of VC could then reduce concentrations to below cleanup levels throughout the plume. Because of low VOC recovery rates, the ability of the pump and treat system to fully remediate contaminated ground water is questionable (Keeley, 1989; U.S. Environmental Protection Agency, 1989; Haley and others, 1991; Freeze and Cherry, 1989; Mackay and Cherry, 1989). The pump and treat system has not had an observable effect on reducing contaminant concentrations in the southern plume—the magnitude and extent of contamination in 1995 and 1998 are essentially identical. Extraction wells have removed some VC from the aquifer since 1995, but the total VC mass removed by the entire pump and treat system through 1998 was only 0.01 kg.

Natural attenuation alone would not immediately prevent all contaminant migration across the southern site boundary. Given the observed 1998 concentrations of VC in the southern plume at the site boundary, the contaminants could migrate across that boundary if the extraction wells were not influencing ground-water flow. However, there is evidence that VC mineralization limited expansion of the southern plume prior to pump and treat, and the VC fate and transport simulations indicate limited expansion of the southern plume if pump and treat were stopped. It is unlikely that natural attenuation processes have ever allowed or will ever allow the plume to extend beyond the boundaries of the Oak Harbor landfill. The effectiveness of the pump and treat system at preventing further contaminant migration in the southern plume is difficult to ascertain from trends in contaminant concentrations alone. The extraction wells PW-2, PW-4, PW-7, PW-8, and PW-9 were designed to control offsite migration of VC in particular, but available chemistry data alone do not confirm that expected performance. VC continues to be detected regularly at well 6-S-29 near the Site boundary, and it continues to not be detected in the downgradient well MW-3B. Those conditions were observed before pump and treat began. A capture-zone analysis of the pump and treat system (Foster Wheeler Environmental, 1998a) did suggest that hydraulic containment was being maintained by the extraction wells. However, extraction wells PW-8 and PW-9 in particular have not been operating consistently since 1995 because of substantial operational problems related to biofouling (Hart Crowser, 1999a).

Given the limited downgradient extent of VC migration in the southern plume, risks to current and future downgradient ground-water users are greatly reduced by natural attenuation. The VC fate and transport simulation established that it is likely that VC concentrations in ground water downgradient of the former Oak Harbor landfill would remain nondetectable and less than cleanup levels due to VC mineralization. There are currently no water-supply wells that penetrate the Oak Harbor landfill, and it is highly improbable that future wells would be installed there because of Washington State laws prohibiting new drinking-water wells within 1,000 ft of a landfill.

Western Contaminant Plume

Natural attenuation alone would not currently be effective at meeting remediation objectives for the western contaminant plume. The pump and treat system currently (1998) appears to be more effective at removing TCA and TCE from shallow ground water. Without the pump and treat system, the existing plume would spread southward from the site boundary. Neither natural attenuation nor pump and treat have prevented all migration of VOC's across the site boundary, particularly to the west of the Navy property.

Natural attenuation has apparently been responsible for some reduction in contaminant concentrations and mass in the western plume, although concentrations of most chlorinated VOC's have exceeded cleanup levels at multiple locations both before and after pump and treat. The fate and transport simulations for TCA suggested that about 1,500 kg of dissolved TCA was degraded over the 15-year period 1980–94, and that peak 1994 concentrations would have been about four times higher without degradation. Data for the pump and treat system shows that 815 kg of TCA has been extracted from the system over the 4-year period 1995–98, so the annual rate of TCA mass removal by pump and treat has exceeded the rate of removal by degradation. About 220 kg of TCE was degraded over the 10-year period 1985–94, and peak 1994 concentrations of TCE would have been about three times higher without degradation. Given that 135 kg of TCE had been extracted from the pump and treat system over the 4-year period 1995–98, the rate of TCE mass removal by pump and treat has also exceeded the rate of removal by degradation. Although the mass of daughter products DCE, DCA, and *cis*DCE in the western plume has also been decreasing, a quantitative evaluation of mass removal by pump and treat compared to biodegradation was not made. Redox conditions in the western plume are favorable for direct mineralization of the dichlorinated compounds, but direct evidence for that biodegradation is not available.

Given that TCA, TCE, and DCE concentrations already exceed cleanup standards at the southern site boundary, immediate cessation of pump and treat would allow the existing western plume to spread southward beyond the site boundary. If pump and treat were discontinued and contaminant degradation continued at the rate estimated for the on-base part of the plume, cleanup standards for at least TCA and DCE would likely be exceeded in a few years at downgradient locations. That would increase the risk to potential ground-water users.

The pump and treat system has had a noticeable positive effect on containing contaminant migration in the western plume. The highest 1995–98 TCA concentrations were at the southwest corner of the Navy property (wells 6-S-25 and PW-5), but TCA concentrations downgradient of the property line (wells 6-S-27 and 6-S-28) steadily decreased during that same time period. Those data suggest that extraction well PW-5 is eliminating or limiting the southward migration of TCA. Concentrations of TCA near the source area (wells PW-1, 6-S-21, N6-37, and N6-38) declined quickly after the onset of pump and treat, but have somewhat stabilized since about mid-1997. Those data suggest that extraction well PW-1 has removed much of the TCA that was in ground water in 1995, but a continuing source for TCA in ground water is still present. Concentrations of TCA in the western edge of the plume (well 6-S-24) appear to have increased from 1995 through 1997, but have since been declining. Those data suggest that extraction well PW-3 may be limiting TCA migration to the west of the Navy property, but the paucity of off-base shallow wells in the area north of 6-S-24 makes defining the lateral extent of the plume uncertain. Well PW-3 also appears to have intercepted much of the southward migrating TCA from the source area, as evidenced by the more rapid concentration decline in downgradient well 6-S-6. The TCE data show similar trends during 1995–98, except that the highest concentrations of TCE had migrated downgradient to around wells PW-3 and 6-S-6, about 1,000 ft north of the highest TCA concentrations.

CONCLUSIONS

Natural Attenuation as an Alternative to Pump and Treat

Natural attenuation is a viable alternative to pump and treat for meeting remediation objectives in the vicinity of the southern contaminant plume. The combination of historically low contaminant concentrations in ground water, a landfill cap that limits source area contributions, favorable conditions for degradation of VC, and a relatively long downgradient distance to potential receptors are all favorable for natural attenuation as a remediation alternative. Natural attenuation could effectively meet all but one remediation goal that the extraction wells PW-2, PW-4, PW-6, PW-7, PW-8 and PW-9 are currently being employed to meet. The goal of preventing migration of all VC across the site boundary could not be met by natural attenuation. Small amounts of VC would likely migrate south of the site boundary, but the potential for subsequent VC mineralization and existing prohibitions on water-supply wells in the Oak Harbor landfill result in no additional risk to potential receptors. The time frame required for natural attenuation to achieve the remediation goals within the established zone of compliance would be similar to the time frame required for pump and treat to meet those goals; the pump and treat system is containing the migration of VC, but it is neither preventing its formation nor reducing its concentration to any measurable degree.

There would be some possible additional benefits of using natural attenuation as an alternative to pump and treat. The first would be that the amount of treated water that would need to be recharged in the swale north of the landfill would decrease by about half. It has been postulated that the excess recharge water has resulted in a more westerly ground-water flow direction beneath the source area for the western contaminant plume, which results in more off-base migration of contamination across the western site boundary. Less recharge would lessen the impact on the natural north-to-south flow direction in the western plume. The second benefit would be a substantial reduction in the amount of dissolved iron and manganese being extracted from the shallow aquifer and run through the treatment system. Dissolved iron and manganese have been a persistent problem at the Area 6 pump and treat system that has caused occasional shut-downs of the system. The bulk of the iron and manganese has come from the extraction wells PW-2, PW-4, PW-6, PW-7, PW-8 and PW-9 located downgradient of the former Navy landfill. Minimizing system shutdowns would result in more effective containment and removal of contamination in the western contaminant plume.

At this time, natural attenuation is a less clearly viable alternative to pump and treat for meeting remediation objectives in the western contaminant plume. Concentrations of TCA, TCE, and DCE in particular are currently too high to be effectively remediated by natural attenuation alone without increasing the risk to downgradient receptors. There is a possibility that rates for reductive dechlorination of TCE and TCA could increase substantially if the plume was allowed to migrate beneath the former Oak Harbor landfill, but there are not enough data from the few wells beneath that landfill to be certain of such degradation. Assuming the more conservative degradation rates determined used in this evaluation, TCA and TCE concentrations need to be reduced to at least 25 and 40 μg/L, respectively, at the southern site boundary to protect potential downgradient receptors. Source area TCA and TCE concentrations have decreased substantially over the past 10 years, and the extraction wells PW-3 and PW-5 in particular are removing a significant mass of contaminants from ground water, so natural attenuation may become a more viable remediation alternative for the western plume in the near future.

Monitoring Needs for Natural Attenuation

The most critical data gap identified in this evaluation is the paucity of contaminant chemistry information downgradient of the Navy boundary in the vicinity of the southern contaminant plume. Without such data, the behavior of the plume cannot be verified, the protectiveness of natural attenuation to downgradient receptors cannot be verified with certainty, and actual (rather than estimated) field attenuation rates for VC cannot be determined. Two former water-supply wells downgradient of the site have been closed (the AHI wells), and the City of Oak Harbor monitoring well MW-3B may not be located along the flow path of the souther plume. Given that the 129 tons of dry-cleaning waste that were reportedly disposed of in the Oak harbor landfill has never been accounted for, there is a risk of finding contamination from a non-Navy source in any new downgradient wells.

The existing long-term monitoring plan would likely need to be reviewed and revised if natural attenuation is selected as a remedy for the contamination in the southern plume. The U.S. Environmental Protection Agency (EPA) has published some guidelines to help with such an effort (U.S. Environmental Protection Agency, 1997). In particular, additional performance monitoring wells (as described in the previous paragraph) may be required downgradient of the site boundary if monitored natural attenuation is adopted.

Future Investigations Related to Natural Attenuation

A more rigorous evaluation of ground-water flow and contaminant transport under various pumping scenarios at Area 6 would be helpful for siting new monitoring wells, and for evaluating changes in ground-water flow that would result from cessation of selected monitoring wells. The cessation of pumping in the southern plume would likely lead to Navy landfill leachate flowing more westwardly, resulting in more strongly reducing redox conditions near the leading edge of the western contaminant plume. Such a change could lead to more favorable conditions for reductive dechlorination of TCA and TCE downgradient of the site boundary. Periodic monitoring of redox-sensitive constituents in Oak Harbor landfill wells would be useful to identify such changes.

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