

USER'S GUIDE FOR POLYETHYLENE-BASED PASSIVE DIFFUSION BAG SAMPLERS TO OBTAIN VOLATILE ORGANIC COMPOUND CONCENTRATIONS IN WELLS

PART 1: DEPLOYMENT, RECOVERY, DATA INTERPRETATION, AND QUALITY CONTROL AND ASSURANCE

Water-Resources Investigations Report 01-4060

PART 2: FIELD TESTS

Water-Resources Investigations Report 01-4061

Prepared in cooperation with the

U.S. AIR FORCE

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U.S. ENVIRONMENTAL PROTECTION AGENCY

FEDERAL REMEDIATION TECHNOLOGIES ROUNDTABLE

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User's Guide for Polyethylene-Based Passive Diffusion Bag Samplers to Obtain Volatile Organic Compound Concentrations in Wells

Part 1: Deployment, Recovery, Data Interpretation, and Quality Control and Assurance

By Don A. Vroblesky

U.S. Geological Survey

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Conversion Factors, Vertical Datum, Acronyms, and Abbreviations

Multiply	By	To obtain
<i>Length</i>		
inch (in.)	25.4	millimeter
foot (ft)	0.3048	meter
mile (mi)	1.609	kilometer
<i>Area</i>		
square mile (mi ²)	2.590	square kilometer
<i>Flow</i>		
foot per day (ft/d)	0.3048	meter per day
foot squared per day (ft ² /d)	0.09294	meter squared per day
gallon per minute (gal/min)	0.06308	liter per second
gallon per day (gal/d)	0.003785	cubic meter per day
inch per year (in/yr)	25.4	millimeter per year
<i>Volume</i>		
gallon (gal)	3.785	liter

Temperature is given in degrees Celsius (°C), which can be converted to degrees Fahrenheit (°F) by the following equation: °F = 9/5 (°C) + 32

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

Chemical concentration in water is expressed in metric units as milligrams per liter (mg/L) or micrograms per liter (µg/L).

Additional Abbreviations	
EDB	1,2-Dibromomethane
AFCEE	Air Force Center for Environmental Excellence
cDCE	<i>cis</i> -1,2-Dibromoethene
ft ³ /d	cubic feet per day
ft ³ /mg	cubic feet per milligram
°C	degrees Celsius
g	gram
ITRC	Interstate Technology Regulatory Cooperation
LDPE	low-density polyethylene
L	liter
µg	microgram
µm	micrometer
µL	microliter
mg	milligram
mL	milliliter
mL/min	milliliter per minute
MTBE	Methyl- <i>tert</i> -butyl ether
NAVFAC	Naval Facilities Engineering Command
NAPL	non-aqueous phase liquid
PDB	passive diffusion bag
PCE	Tetrachloroethene
TCE	Trichloroethene
USEPA	U.S. Environmental Protection Agency
USGS	U.S. Geological Survey
VOA	Volatile organic analysis
VOC	Volatile organic compound

User's Guide for Polyethylene-Based Passive Diffusion Bag Samplers to Obtain Volatile Organic Compound Concentrations in Wells

Part 1: Deployment, Recovery, Data Interpretation, and Quality Control and Assurance

By Don A. Vroblesky

EXECUTIVE SUMMARY

Water-filled passive diffusion bag (PDB) samplers described in this report are suitable for obtaining concentrations of a variety of volatile organic compounds (VOCs) in ground water at monitoring wells. The suggested application of the method is for long-term monitoring of VOCs in ground-water wells at well-characterized sites.

The effectiveness of the use of a single PDB sampler in a well is dependent on the assumption that there is horizontal flow through the well screen and that the quality of the water is representative of the ground water in the aquifer directly adjacent to the screen. If there are vertical components of intra-bore-hole flow, multiple intervals of the formation contributing to flow, or varying concentrations of VOCs vertically within the screened or open interval, then a multiple deployment of PDB samplers within a well may be more appropriate for sampling the well.

A typical PDB sampler consists of a low-density polyethylene (LDPE) lay-flat tube closed at both ends and containing deionized water. The sampler is positioned at the target horizon of the well by attachment to a weighted line or fixed pipe.

The amount of time that the sampler should be left in the well prior to recovery depends on the time required by the PDB sampler to equilibrate with ambient water and the time required for the environmental disturbance caused by sampler deployment to return to ambient conditions. The rate that the water within the PDB sampler equilibrates with ambient water depends on multiple factors, including the type of compound being sampled and the water temperature. The concentrations of benzene, *cis*-1,2-dichloroethene,

tetrachlorethene, trichloroethene, toluene, naphthalene, 1,2-dibromoethane, and total xylenes within the PDB samplers equilibrated with the concentrations in an aqueous mixture of those compounds surrounding the samplers under laboratory conditions within approximately 48 hours at 21 degrees Celsius (°C). A subsequent laboratory study of mixed VOCs at 10 °C showed that tetrachloroethene and trichloroethene were equilibrated by about 52 hours, but other compounds required longer equilibration times. Chloroethane, *cis*-1,2-dichloroethene, *trans*-1,2-dichloroethene, and 1,1-dichloroethene were not equilibrated at 52 hours, but appeared to be equilibrated by the next sampling point at 93 hours. Vinyl chloride, 1,1,1-trichloroethane, 1,2-dichloroethane, and 1,1-dichloroethane were not equilibrated at 93 hours, but were equilibrated by the next sampling point at 166 hours. Different equilibration times may exist for other compounds. Differences in equilibration times, if any, between single-solute or mixed VOC solutions have not yet been thoroughly examined.

The samplers should be left in place long enough for the well water, contaminant distribution, and flow dynamics to restabilize following sampler deployment. Laboratory and field data suggest that 2 weeks of equilibration probably is adequate for many applications; therefore, a minimum equilibration time of 2 weeks is suggested. In less permeable formations, longer equilibration times may be required. When applying PDB samplers in waters colder than previously tested (10 °C) or for compounds without sufficient corroborating data, a side-by-side comparison with conventional methodology is advisable to justify the field equilibration time.

Following the initial equilibration period, the samplers maintain equilibrium concentrations with the ambient water until recovery. Thus, there is no specified time for sampler recovery after initial equilibration. PDB samplers routinely have been left in ground waters having concentrations of greater than 500 parts per million (ppm) of trichloroethene for 3 months at a time with no loss of bag integrity, and at one site, the PDB samplers have been left in place in VOC-contaminated ground water for 1 year with no reported loss of sampler integrity. The effects of long-term (greater than 1 month) PDB-sampler deployment on sampler and sample integrity have not yet been thoroughly tested for a broad range of compounds and concentrations, however. Moreover, in some environments, development of a biofilm on the polyethylene may be a consequence of long-term deployment. Investigations of semipermeable membrane devices (SPMDs) have shown that the transfer of some compounds across a heavily biofouled polyethylene membrane may be reduced, but not stopped. If a heavy organic coating is observed on a PDB sampler, it is advisable to determine the integrity of the sample by comparison to a conventional sampling method before continuing to use PDB samplers for long-term deployment in that well.

Recovery consists of removing the samplers from the well and immediately transferring the enclosed water to 40-milliliter sampling vials for analysis. The resulting concentrations represent an integration of chemical changes over the most recent portion of the equilibration period (approximately 48 to 166 hours, depending on the water temperature and the type of compound).

The method has both advantages and limitations when compared to other sampling methods. Advantages include the potential for PDB samplers to eliminate or substantially reduce the amount of purge water associated with sampling. The samplers are relatively inexpensive and easy to deploy and recover. Because PDB samplers are disposable, there is no downhole equipment to be decontaminated between wells, and there is a minimum amount of field equipment required. The samplers also have the potential to delineate contaminant stratification in the formation across the open or screened intervals of monitoring wells where vertical hydraulic gradients are not present. In addition, the samplers integrate concentrations over time, which may range between about 48 to 166 hours depending on the compound of interest. Because the pore size of LDPE is only about

10 angstroms or less, sediment does not pass through the membrane into the bag. Thus, PDB samplers are not subject to interferences from turbidity. In addition, none of the data collected suggest that VOCs leach from the LDPE material, or that there is a detrimental effect on the VOC sample from the PDB material.

Water-filled polyethylene PDB samplers are not appropriate for all compounds. The samplers are not suitable for inorganic ions and have a limited applicability for non-VOCs and for some VOCs. For example, although methyl-*tert*-butyl ether and acetone and most semivolatile compounds are transmitted through the polyethylene bag, laboratory tests have shown that the resulting concentrations were lower than in ambient water. A variety of factors influence the ability of compounds to diffuse through the polyethylene. These factors include the molecular size and shape and the hydrophobic nature of the compound. Unpublished laboratory test data of semivolatile compounds in contact with PDB samplers showed a higher concentration of phthalates inside the PDB sampler than outside the PDB sampler, suggesting that the polyethylene may contribute phthalates to the enclosed water. Thus, the samplers should not be used to sample for phthalates.

VOC concentrations in PDB samplers represent concentrations in the vicinity of the sampler within the well screen or open interval. This may be a limitation for PDB samplers and some other types of sampling, such as low-flow sampling, if the ground-water contamination is above or below the screen or not in the sample intervals providing water movement to the PDB samplers. If there is a vertical hydraulic gradient in the well, then the concentrations in the sampler may represent the concentrations in the water flowing vertically past the sampler rather than in the formation directly adjacent to the sampler. Vertically spaced multiple PDB samplers may be needed in chemically stratified wells or where flow patterns through the screen change as a result of ground-water pumping or seasonal water-level fluctuations.

The purposes of this document are to present methods for PDB sampler deployment, and recovery; to discuss approaches to determine the applicability of passive diffusion samplers; and to discuss various factors influencing interpretation of the data. The intended audience for the methodology sections of this report is managers and field personnel involved in using PDB samplers. The discussion of passive diffusion sampler applicability and interpretation of the data is

suiting for project managers, technical personnel, and the regulatory community. Part 2 of this report presents case studies of PDB sampler field applications.

INTRODUCTION

The use of PDB samplers for collecting ground-water samples from wells offers a cost-effective approach to long-term monitoring of VOCs at well-characterized sites (Vroblesky and Hyde, 1997; Gefell and others, 1999). The effectiveness of the use of a single PDB sampler in a well is dependent on the assumption that there is horizontal flow through the well screen and that the quality of the water is representative of the ground water in the aquifer directly adjacent to the screen. If there are vertical components of intra-borehole flow, multiple intervals of the formation contributing to flow, or varying concentrations of VOCs vertically within the screened or open interval, then deployment of multiple PDB samplers within a well may be more appropriate for sampling the well.

The samplers consist of deionized water enclosed in a LDPE sleeve (fig. 1) and are deployed adjacent to a target horizon within a screened or open interval of a well. The suggested application is for long-term monitoring of VOCs in ground-water wells. Where the screened interval is greater than 10 feet (ft), the potential for contaminant stratification and/or intra-borehole flow within the screened interval is greater than in screened intervals shorter than 10 ft. It is important that the vertical distribution of contaminants be determined in wells having 10-ft-long well screens, and that both the vertical distribution of contaminants and the potential for intra-borehole flow be determined in wells having screens longer than 10 ft. For many VOCs of environmental interest (table 1), the VOC concentration in water within the sampler approaches the VOC concentration in water outside of the PDB sampler over an equilibration period. The resulting concentrations represent an integration of chemical changes over the most recent part of the equilibration period (approximately 48 to 166 hours, depending on the water temperature and the type of compound being sampled). The approach is inexpensive and has the potential to eliminate or substantially reduce the amount of purge water removed from the well.

A variety of PDB samplers have been utilized in well applications (fig. 1). Although the samplers vary in specific construction details, a typical PDB sampler consists of a 1- to 2-ft-long LDPE tube closed at both ends and containing laboratory-grade deionized water (fig. 1). The typical diameter for PDB samplers used in a 2-inch-diameter well is approximately 1.2 inches; however, other dimensions may be used to match the well diameter. Equilibration times may be longer for larger diameter PDB samplers. On the outside of the PDB sampler, a low-density polyethylene-mesh sometimes is used for protection against abrasion in open boreholes and as a means of attachment at the prescribed depth. The PDB sampler can be positioned at the target horizon by attachment to a weighted line or by attachment to a fixed pipe.

PDB samplers for use in wells are available commercially. Authorized distributors as of March 2001 are Columbia Analytical Services (800-695-7222; www.caslab.com) and Eon Products (800-474-2490; www.eonpro.com). A current list of vendors and PDB-sampler construction details can be obtained from the U.S. Geological Survey Technology Transfer Enterprise Office, Mail Stop 211, National Center, 12201 Sunrise Valley Drive, Reston, Virginia 20192 (telephone 703-648-4344; fax 703-648-4408). PDB samplers employ patented technology (U.S. patent number 5,804,743), and therefore, require that the user purchase commercially produced samplers from a licensed manufacturer or purchase a nonexclusive license for sampler construction from the U.S. Geological Survey Technology Enterprise Office at the above address.

The purposes of this document are to present methods for PDB sampler deployment, and recovery; to discuss approaches for determining the applicability of passive diffusion samplers; and to discuss various factors influencing interpretation of the data. The intended audience for the methodology sections of this report is managers and field personnel involved in using PDB samplers. The discussion of PDB sampler applicability and interpretation of the data is suited for project managers, technical personnel, and the regulatory community. Part 2 of this report presents case studies of PDB-sampler field applications.

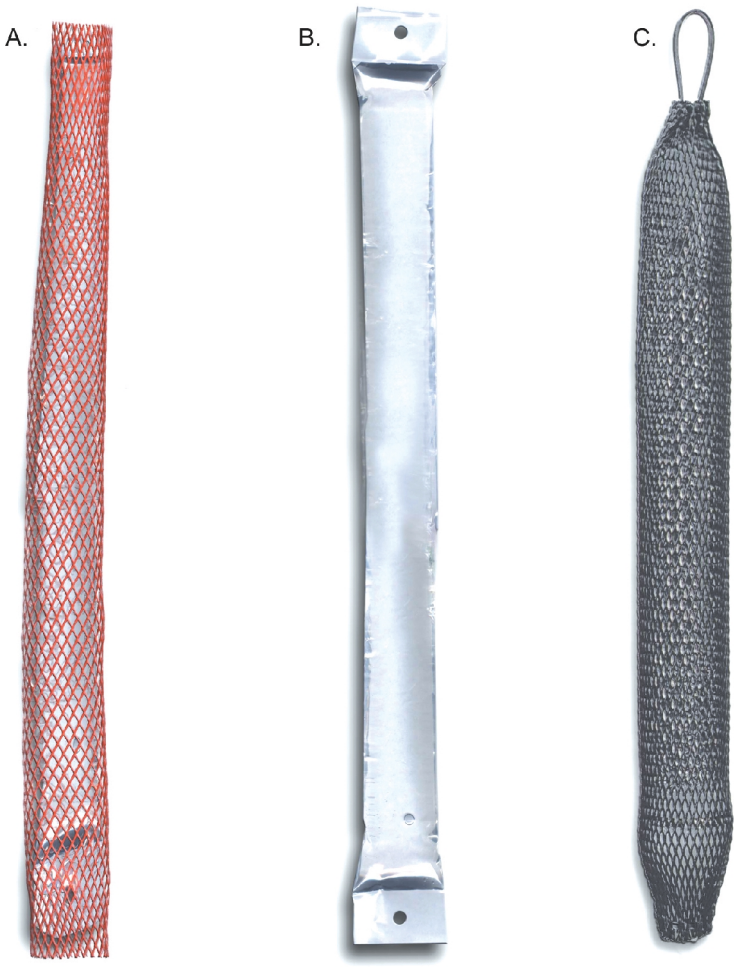


Figure 1. Typical water-filled passive diffusion bag samplers used in wells, including (A) diffusion bag with polyethylene mesh, (B) diffusion bag without mesh, and (C) bag and mesh attached to bailer bottom.

Table 1. Compounds tested under laboratory conditions for use with passive diffusion bag samplers [From Vroblesky and Campbell, 2001]

Tested compounds showing good correlation (average differences in concentration of 11 percent or less between diffusion-sampler water and test-vessel water) in laboratory tests			
Benzene	2 Chlorovinyl ether	<i>cis</i> -1,2-Dichloroethene	1,1,1-Trichloroethane
Bromodichloromethane	Dibromochloromethane	<i>trans</i> -1,2-Dichloroethene	1,1,2-Trichloroethane
Bromoform	Dibromomethane	1,2-Dichloropropane	Trichloroethene
Chlorobenzene	1,2-Dichlorobenzene	<i>cis</i> -Dichloropropene	Trichlorofluoromethane
Carbon tetrachloride	1,3-Dichlorobenzene	1,2-Dibromoethane	1,2,3-Trichloropropane
Chloroethane	1,4-Dichlorobenzene	<i>trans</i> -1,3-Dichloropropene	1,1,2,2-Tetrachloroethane
Chloroform	Dichlorodifluoromethane	Ethyl benzene	Tetrachloroethene
Chloromethane	1,2-Dichloroethane	Naphthalene	Vinyl chloride
	1,1-Dichloroethene	Toluene	Total xylenes
Tested compounds showing poor correlation (average differences in concentration greater than 20 percent between diffusion-sampler water and test-vessel water) in laboratory tests			
Acetone*	Methyl- <i>tert</i> -butyl ether	Styrene	

*T.M Sivavec and S.S. Baghel, General Electric Company, written commun., 2000

Summary of Passive Diffusion Bag Sampler Advantages and Limitations

Advantages

1. PDB samplers have the potential to eliminate or substantially reduce the amount of purge water associated with sampling.

2. PDB samplers are inexpensive.

3. The samplers are easy to deploy and recover.

4. Because PDB samplers are disposable, there is no downhole equipment to be decontaminated between wells.

5. A minimal amount of field equipment is required.

6. Sampler recovery is rapid. Because of the small amount of time and equipment required for the sampling event, the method is practical for use where access is a problem or where discretion is desirable (that is, residential communities, business districts, or busy streets where vehicle traffic control is a concern).

7. Multiple PDB samplers, distributed vertically along the screened or open interval, may be used in conjunction with borehole flow meter testing to gain insight on the movement of contaminants into and out of the well screen or open interval or to locate the zone of highest concentration in the well. Analytical costs when using multiple PDB samplers sometimes can be reduced by selecting a limited number of the samplers for laboratory analysis based on screening by using field gas chromatography at the time of sample collection.

8. Because the pore size of LDPE is only about 10 angstroms or less, sediment does not pass through the membrane into the bag. Thus, PDB samplers are not subject to interferences from turbidity. In addition, none of the data collected suggest that VOCs leach from the LDPE material or that there is a detrimental effect from the PDB material on the VOC sample.

Limitations

1. PDB samplers integrate concentrations over time. This may be a limitation if the goal of sampling is to collect a representative sample at a point in time in an aquifer where VOC-concentrations substantially change more rapidly than the samplers equilibrate. Laboratory results obtained indicate that a variety of compounds equilibrated within 48 hours at 21 °C (Vroblesky and Campbell, 2001). Vinyl chloride, 1,1,1-trichloroethane, 1,2-dichloroethane, and 1,1-dichloroethane may require between 93 and 166 hours to equilibrate at 10 °C (T.M. Sivavec and S.S. Baghel, General Electric Company, written commun., 2000). The initial equilibration under field conditions may be longer to allow

well water, contaminant distribution, and flow dynamics to restabilize following sampler deployment.

2. Water-filled polyethylene PDB samplers are not appropriate for all compounds. For example, although methyl-*tert*-butyl ether and acetone (Vroblesky, 2000; Paul Hare, General Electric Company, oral commun., 2000) and most semivolatile compounds are transmitted through the polyethylene bag, laboratory tests have shown that the resulting concentrations were lower than in ambient water. A variety of factors influence the ability of compounds to diffuse through the polyethylene membrane. These factors include the molecular size and shape and the hydrophobic nature of the compound. Compounds having a cross-sectional diameter of about 10 angstroms or larger (such as humic acids) do not pass through the polyethylene because the largest (transient) pores in polyethylene do not exceed about 10 angstroms in diameter (Flynn and Yalkowsky, 1972; Hwang and Kammermeyer, 1975; Comyn, 1985). The samplers are not appropriate for hydrophilic polar molecules, such as inorganic ions. A detailed discussion of the relation between hydrophobicity and compound transport through polyethylene can be found in Gale (1998). Unpublished laboratory test data (D.A. Vroblesky, U.S. Geological Survey, written commun., 1998) of semi-volatile compounds in contact with PDB samplers showed a higher concentration of phthalates inside the PDB sampler than outside the PDB sampler, suggesting that the polyethylene may contribute phthalates to the enclosed water. Thus, the samplers should not be used to sample for phthalates.

3. PDB samplers rely on the free movement of water through the well screen. In situations where ground water flows horizontally through the well screen, the VOC concentrations in the open interval of the well probably are representative of the aquifer water in the adjacent formation (Gillham and others, 1985; Robin and Gillham, 1987; Kearl and others, 1992; Powell and Puls, 1993; Vroblesky and Hyde, 1997). In these situations, the VOC concentration of the water in contact with the PDB samplers, and therefore, the water within the diffusion samplers, probably represents local conditions in the adjacent aquifer. However, if the well screen is less permeable than the aquifer or the sandpack, then under ambient conditions, flowlines may be diverted around the screen. Such a situation may arise from inadequate well development or from iron bacterial fouling of the well screen. In this case, the VOC concentrations in the PDB samplers may not represent concentrations in

the formation water because of inadequate exchange across the well screen. PDB samplers have not yet been adequately tested to determine their response under such conditions.

4. VOC concentrations in PDB samplers represent ground-water concentrations in the vicinity of the screened or open well interval that move to the sampler under ambient flow conditions. This is a limitation if the ground-water contamination lies above or below the well screen or open interval, and requires the operation of a pump to conduct contaminants into the well for sampling.

5. In cases where the well screen or open interval transects zones of differing hydraulic head and variable contaminant concentrations, VOC concentrations obtained using a PDB sampler may not reflect the concentrations in the aquifer directly adjacent to the sampler because of vertical transport in the well. However, a vertical array of PDB samplers, used in conjunction with borehole flow meter testing, can provide insight on the movement of contaminants into or out of the well. This information then can be used to help determine if the use of PDB samplers is appropriate for the well, and to select the optimal vertical location(s) for the sampler deployment.

6. In wells with screens or open intervals with stratified chemical concentrations, the use of a single PDB sampler set at an arbitrary (by convention) depth may not provide accurate concentration values for the most contaminated zone. However, multiple PDB samplers distributed vertically along the screened or open interval, in conjunction with pump sampling (as appropriate), can be used to locate zone(s) of highest concentration in the well. Multiple PDB samplers also may be needed to track the zone of maximum concentration in wells where flow patterns through the screened interval change as a result of ground-water pumping or seasonal water-table fluctuations.

PASSIVE DIFFUSION BAG SAMPLER DEPLOYMENT

A variety of approaches can be used to deploy the PDB samplers in wells. A typical deployment approach, described in this section, is to attach the PDB samplers to a weighted line. It also is acceptable to attach the weights directly to the PDB sampler if the attachment point is of sufficient strength to support the weight. The weights attached to the bottom of the

line are stainless steel and can be reused, but must be thoroughly decontaminated with a detergent before the first use or before using in a different well. Rope, such as 90 pound, 3/16 inch braided polyester, can be used as the line for single-use applications if it is of sufficient strength to support the weight and sampler, is nonbuoyant, and is subject to minimal stretch; however, the rope should not be reused because of the high potential for cross contamination. Stainless-steel or Teflon-coated stainless-steel wire is preferable. The weighted lines should not be reused in different wells to prevent carryover of contaminants. A possible exception is coated stainless-steel wire, which can be reused after sufficient decontamination. An alternative deployment approach, not discussed in this section, is to attach the PDB samplers to a fixed pipe in the well (Vroblesky and Peters, 2000, p. 3; also included in Part 2 of this publication). The PDB samplers should not contact non-aqueous phase liquid (NAPL) during deployment or retrieval to prevent cross contamination. An approach that can be utilized to deploy diffusion samplers through a layer of floating NAPL is described in the field test at Naval Station North Island, California (Vroblesky and Peters, 2000, p. 3-4; also included in Part 2 of this publication).

If the PDB sampler is to be compared with a conventional pumping approach to sampling, then it is suggested that both the pump and the PDB sampler be deployed at the same time, with the sampler attached near (such as directly below) the pump inlet. This approach eliminates potential concentration differences between the two methods that may result from well disturbance during equipment removal and deployment at the time of sampling. An alternative method is to deploy the PDB samplers independently of the pumps and recover the samplers immediately prior to placing the pump down the well.

PDB samplers are available either prefilled (field ready) with laboratory-grade deionized water or unfilled. The unfilled samplers are equipped with a plug and funnel to allow for field filling and sample recovery. To fill these samplers, remove the plug from the sampler bottom, insert the short funnel into the sampler, and pour laboratory-grade deionized water into the sampler. The sampler should be filled until water rises and stands at least half way into the funnel. Remove excess bubbles from the sampler. Remove the funnel and reattach the plug. A small air bubble from the plug is of no concern.

The following steps should be used for deploying PDB samplers in wells:

1. Measure the well depth and compare the measured depth with the reported depth to the bottom of the well screen from well-construction records. This is to check on whether sediment has accumulated in the bottom of the well, whether there is a nonscreened section of pipe (sediment sump) below the well screen, and on the accuracy of well-construction records. If there is an uncertainty regarding length or placement of the well screen, then an independent method, such as video imaging of the well bore, is strongly suggested.

2. Attach a stainless-steel weight to the end of the line. Sufficient weight should be added to counterbalance the buoyancy of the PDB samplers. This is particularly important when multiple PDB samplers are deployed. One approach, discussed in the following paragraphs, is to have the weight resting on the bottom of the well, with the line taut above the weight. Alternatively, the PDB sampler and weight may be suspended above the bottom, but caution should be exercised to ensure that the sampler does not shift location. Such shifting can result from stretching or slipping of the line or, if multiple samplers are attached end-to-end rather than to a weighted line, stretching of the samplers.

3. Calculate the distance from the bottom of the well, or top of the sediment in the well, up to the point where the PDB sampler is to be placed. A variety of approaches can be used to attach the PDB sampler to the weight or weighted line at the target horizon. The field-fillable type of PDB sampler is equipped with a hanger assembly and weight that can be slid over the sampler body until it rests securely near the bottom of the sampler. When this approach is used with multiple PDB samplers down the same borehole, the weight should only be attached to the lowermost sampler. An additional option is to use coated stainless-steel wire as a weighted line, making loops at appropriate points to attach the upper and lower ends of PDB samplers. Where the PDB sampler position varies between sampling events, movable clamps with rings can be used. When using rope as a weighted line, a simple approach is to tie knots or attach clasps at the appropriate depths. Nylon cable ties or stainless-steel clips inserted through the knots can be used to attach the PDB samplers. An approach using rope as a weighted line with knots tied at the appropriate sampler-attachment points is discussed below.

- (a) For 5-ft-long or shorter well screens, the center point of the PDB sampler should be the vertical midpoint of the saturated well-screen length. For example, if the well screen is at a depth of 55 to 60 ft below the top of casing, and the measured depth of the well is 59 ft, then the bottom of the well probably has filled with sediment. In this case, the midpoint of the sampler between the attachment points on the line will be midway between 55 and 59 ft, or at 57 ft. Thus, for a 1.5-ft-long sampler, the attachment points on a weighted line should be tied at distances of 1.25 ft ($2 \text{ ft} - 0.75 \text{ ft}$) and 2.75 ft ($2 \text{ ft} + 0.75 \text{ ft}$) from the top of the sediment in the well, or the bottom of the well, making adjustments for the length of the attached weight. When the PDB sampler is attached to the line and installed in the well, the center of the sampler will be at 57-ft depth. If, however, independent evidence is available showing that the highest concentration of contaminants enters the well from a specific zone within the screened interval, then the PDB sampler should be positioned at that interval.

- (b) For 5- to 10-ft-long well screens, it is advisable to utilize multiple PDB samplers vertically along the length of the well screen for at least the initial sampling (fig. 2). The purposes of the multiple PDB samplers are to determine whether contaminant stratification is present and to locate the zone of highest concentration. The midpoint of each sampler should be positioned at the midpoint of the interval to be sampled. For 1.5-ft-long samplers, at each sampling depth in the screened interval, make two attachment points on the weighted line at a distance of about 1.5 ft apart. The attachment points should be positioned along the weighted line at a distance from the bottom end of the weight such that the midpoint between the knots will be at the desired sampling depth along the well screen. Sampler intervals are variable, but a simple approach is to use the top knot/loop of one sampler interval as the bottom knot/loop for the overlying sampler interval.



Figure 2. Example of multiple PDB samplers prepared for deployment.

(c) PDB samplers should not be used in wells having screened or open intervals longer than 10 ft unless used in conjunction with borehole flow meters or other techniques to characterize vertical variability in hydraulic conductivity and contaminant distribution or used strictly for qualitative reconnaissance purposes. This is because of the increased potential for cross contamination of water-bearing zones and hydraulically driven mixing effects that may cause the contaminant stratification in the well to differ from the contaminant stratification in the adjacent aquifer material. If it is necessary to sample such wells, then multiple PDB samplers should be installed vertically across the screened or open interval to determine the zone of highest concentration and whether contaminant stratification is present.

4. The samplers should be attached to the weights or weighted line at the time of deployment. For samplers utilizing the hanger and weight assembly,

the line can be attached directly to the top of the sampler. PDB samplers utilizing an outer protective mesh can be attached to a weighted line by using the following procedure:

(a) Insert cable ties through the attachment points in the weighted line.

(b) At each end of the PDB sampler, weave the ends of the cable ties or clamp through the LPDE mesh surrounding the sampler and tighten the cable ties. Thus, each end of the PDB sampler will be attached to a knot/loop in the weighted line by means of a cable tie or clamp. The cable ties or clamps should be positioned through the polyethylene mesh in a way that prevents the PDB sampler from sliding out of the mesh.

(c) Trim the excess from the cable tie before placing the sampler down the well. Caution should be exercised to prevent sharp edges on the trimmed cable ties that may puncture the LDPE.

5. When using PDB samplers without the protective outer mesh, the holes punched at the ends of the bag, outside the sealed portion, can be used to attach the samplers to the weighted line. Stainless-steel spring clips have been found to be more reliable than cable ties in this instance, but cable ties also work well.

6. Lower the weight and weighted line down the well until the weight rests on the bottom of the well and the line above the weight is taut. The PDB samplers should now be positioned at the expected depth. A check on the depth can be done by placing a knot or mark on the line at the correct distance from the top knot/loop of the PDB sampler to the top of the well casing and checking to make sure that the mark aligns with the lip of the casing after deployment.

7. Secure the assembly in this position. A suggested method is to attach the weighted line to a hook on the inside of the well cap. Reattach the well cap. The well should be sealed in such a way as to prevent surface-water invasion. This is particularly important in flush-mounted well vaults that are prone to flooding.

8. Allow the system to remain undisturbed as the PDB samplers equilibrate.

PASSIVE DIFFUSION BAG SAMPLER AND SAMPLE RECOVERY

The amount of time that the samplers should be left in the well prior to recovery depends on the time required by the PDB sampler to equilibrate with ambient water and the time required for environmental disturbances caused by sampler deployment to return to ambient conditions. The rate that the water within the PDB sampler equilibrates with ambient water depends on multiple factors, including the type of compound being sampled and the water temperature. The concentrations of benzene, *cis*-1,2-dichloroethene (*c*DCE), tetrachlorethene (PCE), trichloroethene (TCE), toluene, naphthalene, 1,2-dibromoethane (EDB), and total xylenes within the PDB samplers equilibrated with the concentrations in an aqueous mixture of those compounds surrounding the samplers under laboratory conditions within approximately 48 hours at 21 °C (Vroblesky and Campbell, 2001). A subsequent laboratory study of mixed VOCs at 10 °C showed that PCE and TCE were equilibrated by about 52 hours, but other compounds required longer equilibration times (T.M. Sivavec and S.S. Baghel, General Electric Company, written commun., 2000). Chloroethane, *c*DCE, *trans*-1,2-dichloroethene, and 1,1-dichloroethene were not

equilibrated at 52 hours, but appeared to be equilibrated by the next sampling point at 93 hours. Vinyl chloride, 1,1,1-trichloroethane, 1,2-dichloroethane, and 1,1-dichloroethane were not equilibrated at 93 hours, but were equilibrated by the next sampling point at 166 hours. Different equilibration times may exist for other compounds. Differences in equilibration times, if any, between single-solute or mixed-VOC solutions have not yet been thoroughly examined.

Under field conditions, the samplers should be left in place long enough for the well water, contaminant distribution, and flow dynamics to restabilize following sampler deployment. The results of borehole dilution studies show that wells can recover to 90 percent of the predisturbance conditions within minutes to several hours for permeable to highly permeable geologic formations, but may require 100 to 1,000 hours (4 to 40 days) in muds, very fine-grained loamy sands, and fractured rock, and may take even longer in fractured shales, recent loams, clays, and slightly fractured solid igneous rocks (Halevy and others, 1967).

In general, where the rate of ground-water movement past a diffusion sampler is high, equilibration times through various membranes commonly range from a few hours to a few days (Mayer, 1976; Harrington and others, 2000). One field investigation showed adequate equilibration of PDB samplers to aquifer trichloroethene (TCE) and carbon tetrachloride (CT) concentrations within 2 days in a highly permeable aquifer (Vroblesky and others, 1999). In other investigations, PDB samplers recovered after 14 days were found to be adequately equilibrated to chlorinated VOCs (O'Brien & Gere Engineers, Inc., 1997a, 1997b; Hare, 2000); therefore, the equilibration period was less than or equal to 14 days for those field conditions. Because it appears that 2 weeks of equilibration probably is adequate for many applications, a minimum equilibration time of 2 weeks is suggested. When applying PDB samplers in waters colder than previously tested (10 °C) or for compounds without sufficient corroborating field data, a side-by-side comparison with conventional sampling methodology is advisable to justify the field equilibration time.

In less permeable formations, longer equilibration times may be required. It is probable that water in the well bore eventually will equilibrate with the pore-water chemistry; however, if the rate of chemical change or volatilization loss in the well bore exceeds the rate of exchange between the pore water and the well-bore water, then the PDB samplers may under-

estimate pore-water concentrations. Guidelines for equilibration times and applicability of PDB samplers in low-permeability formations have not yet been established. Therefore, in such situations, a side-by-side comparison of PDB samplers and conventional sampling methodology is advisable to ensure that the PDB samplers do not underestimate concentrations obtained by the conventional method. A detailed discussion of diffusion rates relevant to diffusion sampler equilibrium in slow-moving ground-water systems can be found in Harrington and others (2000).

Following the initial equilibration period, the samplers maintain equilibrium concentrations with the ambient water until recovery. Thus, there is no specified maximum time for sampler recovery. PDB samplers have routinely been left in ground waters having concentrations of greater than 500 ppm of TCE for 3 months at a time with no loss of bag integrity, and at one site, the PDB samplers have been left in place in VOC-contaminated ground water for 1 year with no reported loss of sampler integrity (Paul Hare, General Electric Company, oral commun., 2000). The effects of long-term (greater than 1 month) PDB-sampler deployment on sampler and sample integrity have not yet been thoroughly tested for a broad range of compounds and concentrations. Moreover, in some environments, development of a biofilm on the polyethylene may be a consequence of long-term deployment. Investigations of semipermeable membrane devices (SPMDs) have shown that the transfer of some compounds may be reduced, but not stopped, across a heavily biofouled polyethylene membrane (Ellis and others, 1995; Huckins and others, 1996; Huckins and others, in press). If a heavy organic coating is observed on a PDB sampler, it is advisable to determine the integrity of the sample by comparing contaminant concentrations from the PDB sampler to concentrations from a conventional sampling method before continuing to use PDB samplers for long-term deployment in that well.

Recovery of PDB samplers is accomplished by using the following approach:

1. Remove the PDB samplers from the well by using the attached line. The PDB samplers should not be exposed to heat or agitated.

2. Examine the surface of the PDB sampler for evidence of algae, iron or other coatings, and for tears in the membrane. Note the observations in a sampling field book. If there are tears in the membrane, the

sample should be rejected. If there is evidence that the PDB sampler exhibits a coating, then this should be noted in the validated concentration data.

3. Detach and remove the PDB sampler from the weighted line. Remove the excess liquid from the exterior of the bag to minimize the potential for cross contamination.

4. A variety of approaches may be used to transfer the water from the PDB samplers to 40-mL volatile organic analysis (VOA) vials. One type of commercially available PDB sampler provides a discharge device that can be inserted into the sampler. If discharge devices are used, the diameter of the opening should be kept to less than about 0.15 inches to reduce volatilization loss. Two options are presently available to recover water from the sample using discharge devices. One option involves removing the hanger and weight assembly from the sampler, inverting the sampler so that the fill plug is pointed upward, and removing the plug. The water can be recovered by directly pouring in a manner that minimizes agitation or by pouring through a VOC-discharge accessory inserted in place of the plug. The second approach involves piercing the sampler near the bottom with a small-diameter discharge tube and allowing water to flow through the tube into the VOA vials. In each case, flow rates can be controlled by tilting or manipulating the sampler. Alternatively, the PDB sampler can be cut open at one end using scissors or other cutting devices which have been decontaminated between use for different wells. Water can then be transferred to 40-ml VOA vials by gently pouring in a manner that minimizes water agitation. Acceptable duplication has been obtained using each method. Preserve the samples according to the analytical method. The sampling vials should be stored at approximately 4 °C in accordance with standard sampling protocol. Laboratory testing suggests that there is no substantial change in the VOC concentrations in PDB samplers over the first several minutes after recovery; however, the water should be transferred from the water-filled samplers to the sample bottles immediately upon recovery.

5. A cost-effective alternative when using multiple PDB samplers in a single well is to field screen water from each sampler using gas chromatography. These results can be used to decide which of the multiple PDB samplers should be sent to an EPA-approved laboratory for standard analysis. Typically, at least the sample containing the highest concentration should be analyzed by a laboratory.

6. If a comparison is being made between concentrations obtained using PDB samplers and concentrations obtained using a conventional sampling approach, then the well should be sampled by the conventional approach soon after (preferably on the same day) recovery of the PDB sampler. The water samples obtained using PDB samplers should be sent in the same shipment, as the samples collected by the conventional approach for the respective wells. Utilizing the same laboratory may reduce analytical variability.

7. Any unused water from the PDB sampler and water used to decontaminate cutting devices should be disposed in accordance with local, state, and Federal regulations.

DETERMINING APPLICABILITY OF PASSIVE DIFFUSION BAG SAMPLERS AND INTERPRETATION OF DATA

When attempting to determine whether the use of PDB samplers is appropriate at a particular well, a common approach is to do a side-by-side comparison with a conventional sampling method during the same sampling event. This approach is strongly suggested in wells having temporal concentration variability. In a well having relatively low temporal concentration variability, comparison of the PDB-sampler results to historical concentrations may provide enough information to determine whether the PDB samplers are appropriate for the well. In general, if both PDB and conventional sampling produce concentrations that agree within a range deemed acceptable by local, state, and Federal regulatory agencies and meet the site-specific data-quality objectives, then a PDB sampler may be approved for use in that well to monitor ambient VOC concentrations. If concentrations from the PDB sampler are higher than concentrations from the conventional method, it is probable that concentrations from the PDB sampler adequately represent ambient conditions because there usually is a greater potential for dilution from mixing during sampling using conventional methods than during sampling using PDB samplers.

If, however, the conventional method produces concentrations that are significantly higher than those obtained using the PDB sampler, then it is uncertain whether the PDB-sampler concentrations represent local ambient conditions. In this case, further testing can be done to determine whether contaminant stratification and/or intra-borehole flow is present. Multiple sampling devices can be used to determine the pres-

ence of contaminant stratification, and borehole flowmeters can be used to determine whether intra-borehole flow is present. When using flowmeters to measure vertical flow in screened boreholes, however, the data should be considered qualitative because of the potential for water movement through the sand pack. Borehole dilution tests (Halevy and others, 1967; Drost and others, 1968; Grisak and others, 1977; Palmer, 1993) can be used to determine whether water is freely exchanged between the aquifer and the well screen.

Once the source of the difference between the two methods is determined, a decision can be made regarding the well-specific utility of the PDB samplers. Tests may show that VOC concentrations from the PDB samplers adequately represent local ambient conditions within the screened interval despite the higher VOC concentration obtained from the conventional method. This may be because the pumped samples incorporated water containing higher concentrations either from other water-bearing zones induced along inadequate well seals or through fractured clay (Vroblesky and others, 2000), from other water-bearing zones not directly adjacent to the well screen as a result of well purging prior to sampling (Vroblesky and Petkewich; 2000), or from mixing of chemically stratified zones in the vicinity of the screened interval (Vroblesky and Peters, 2000).

The mixing of waters from chemically stratified zones adjacent to the screened interval during pumping probably is one of the more important sources of apparent differences between the results obtained from PDB sampling and conventional sampling because such stratification probably is common. Vertical stratification of VOCs over distances of a few feet has been observed in aquifer sediments by using multilevel sampling devices (Dean and others, 1999; Pitkin and others, 1999), and considerable variation in hydraulic conductivity and water chemistry has been observed in an aquifer in Cape Cod, Massachusetts, on the scale of centimeters (Wolf and others, 1991; Smith and others 1991; Hess and others, 1992). Multiple PDB samplers have been used to show a change in TCE concentration of 1,130 ($\mu\text{g/L}$) over a 6-ft vertical screened interval in Minnesota (Vroblesky and Petkewich, 2000). Tests using PDB samplers in screened intervals containing VOC stratification showed that the PDB-sampler data appeared to be point-specific, whereas the pumped sample integrated water over a larger interval (Vroblesky and Peters, 2000).

The decision on whether to use PDB samplers in such situations depends on the data-quality objectives for the particular site. If the goal is to determine and monitor higher concentrations or to examine contaminant stratification within the screened interval, then the PDB samplers may meet this objective. If the goal is to determine the average concentrations for the entire screened interval, then a pumped sample or an average from multiple diffusion samplers may be appropriate.

As an aid in the decision-making process, the following section examines the influences that hydraulic and chemical heterogeneity of an aquifer can have on sample quality in long-screened wells. Because VOC concentrations from PDB samplers commonly are compared to VOC concentrations from other sampling methodologies, the second section examines the differences in sample quality between these methodologies in situations of hydraulic and chemical heterogeneity.

Influences of Hydraulic and Chemical Heterogeneity on Sample Quality in Long-Screened Wells

Sampling biases and chemical variability in long-screened wells, which can be loosely defined as wells having significant physical and chemical heterogeneity within the screened interval and in the adjacent aquifer (Reilly and Leblanc, 1998), have been the subject of numerous investigations. Sources of chemical variability in such wells include non-uniform flow into wells (Robbins and Martin-Hayden, 1991; Reilly and Gibs, 1993; Chiang and others, 1995; Church and Granato, 1996; Reilly and LeBlanc, 1998), lithologic heterogeneity (Reilly and others, 1989; Robbins, 1989; Martin-Hayden and others, 1991; Gibs and others, 1993; Reilly and Gibs, 1993), and in-well mixing. In a well open across a chemically or hydraulically heterogeneous section of the aquifer, differences in the sampling methodology can produce significant differences in the sampling results.

Long-screened wells have the potential to redistribute chemical constituents in the aquifer where there are vertical hydraulic gradients within the screened interval. Water can move into the well from one horizon and exit the well at a different horizon (Church and Granato, 1996; Reilly and LeBlanc 1998). If there is vertical flow in the screened or open interval, and the zone of low hydraulic head (outflow from

the well) is within the contaminated horizon, then the PDB samplers (or any standard sampling methodology) can underestimate or not detect the contamination. The reason is that, in this case, the contaminated horizon does not contribute water to the well under static conditions. Instead, water from other horizons with higher hydraulic head will invade the contaminated horizon by way of the well screen. Under pumped conditions, the majority of the extracted water will be from the most permeable interval, which may not be the contaminated zone. Even when pumping induces inflow from the contaminated interval, much of that inflow will be a reflection of the residual invaded water from other horizons. In this situation, a substantial amount of purging would be required before water representative of the aquifer could be obtained (Jones and Lerner, 1995). Such sampling is not likely to reflect a significant contribution from the contaminated zone, and concentrations in the contaminated zone probably will be underestimated.

Similarly, if VOC-contaminated water is flowing into the well and is exiting the well at a different horizon, then VOCs will be present along the screened interval between the two horizons. In this case, VOC concentrations in the screened interval may be representative of aquifer concentrations at the inflow horizon, but may not be representative of aquifer concentrations near the outflow horizon.

In areas where vertical stratification of VOC concentrations is anticipated, using multiple PDB samplers may more fully characterize the contaminated horizon than using a single PDB sampler. This is particularly true in wells having screens 10 ft or longer; however, significant VOC stratification has been observed over intervals of less than 5 ft (Vroblesky and Peters, 2000). Because of the increased probability of vertical concentration or hydraulic gradients within the open interval of long-screened (greater than 10 ft) wells, it is advisable to determine the zones of inflow and outflow within the screened or open interval of these wells using borehole flowmeter analysis (Hess, 1982; 1984; 1986; 1990; Young and others, 1998).

Comparison of Passive Diffusion Bag Sampling Methodology to Conventional Methodologies

Traditional sampling methodologies, such as the purge-and-sample (or conventional purging method), low-flow or low-volume sampling, and using straddle packers and multilevel samplers, produce VOC

concentrations that may differ from VOC concentrations obtained from PDB samplers because the methodologies sometimes are influenced in different ways by aquifer hydraulic and chemical heterogeneity. This section examines potential sources of concentration differences between traditional methodologies and the PDB methodology.

The purge-and-sample approach to ground-water monitoring differs from the diffusion-sampler approach primarily because the area of the screened or open interval that contributes water to the purged sample typically is greater than for the PDB sampler, and the potential for mixing of stratified layers is higher. When pumping three or more casing volumes of water prior to collecting a sample, chemical concentrations in the discharging water typically change as the well is pumped (Keely and Boateng, 1987; Cohen and Rabold, 1988; Martin-Hayden and others, 1991; Robbins and Martin-Hayden, 1991; Reilly and Gibs, 1993; Barcelona and others, 1994; Martin-Hayden, 2000), due to mixing during pumping and other factors, such as the removal of stagnant water in the casing and changing patterns of inflow and outflow under ambient and pumping conditions (Church and Granato, 1996). The induction of lateral chemical heterogeneity during pumping also may produce variations in the sampled concentrations. The amount of mixing during purging can be highly variable (Barber and Davis, 1987; Church and Granato, 1996; Reilly and LeBlanc, 1998; Martin-Hayden, 2000), and may result in concentrations that are not locally representative (Reilly and Gibs, 1993). Substantial vertical hydraulic gradients, even in shallow homogeneous aquifers, have been observed to bias sampling using conventional purging because the majority of the pumped water may come from a particular horizon not related to the contaminated zone and because the intra-well flow that intruded the aquifer may not be adequately removed during purging (Hutchins and Acree, 2000). Thus, differences may be observed between concentrations obtained from a pumped sample and from a PDB sample in a chemically stratified interval if the pumped sample represents an integration of water collected from multiple horizons and the PDB sampler represents water collected from a single horizon.

Low-flow purging and sampling (Barcelona and others, 1994; Shanklin and others, 1995) disturbs the local ground water less than conventional purge-and-

sample methods. Thus, samples obtained by PDB samplers are likely to be more similar to samples obtained by using low-flow purging than to those obtained by using conventional purge-and-sample methods. Even under low-flow conditions, however, purging still can integrate water within the radius of pumping influence, potentially resulting in a deviation from VOC concentrations obtained by PDB sampling. One investigation found that in low hydraulic conductivity formations, low-flow sampling methodology caused excessive drawdown, which dewatered the screened interval, increased local ground-water velocities, and caused unwanted colloid and soil transport into the ground-water samples (Sevee and others, 2000). The authors suggest that in such cases, a more appropriate sampling methodology may be to collect a slug or passive sample from the well screen under the assumption that the water in the well screen is in equilibrium with the surrounding aquifer.

Isolating a particular contributing fracture zone with straddle packers in an uncased borehole allows depth-discrete samples to be collected from the target horizon (Hsieh and others, 1993; Kaminsky and Wylie, 1995). Strategically placed straddle packers often can minimize or eliminate the impact of vertical gradients in the sampled interval. However, even within a packed interval isolating inflowing fracture zones, deviations between VOC concentrations in water from PDB samplers and water sampled by conventional methods still may occur if the conventional method mixes chemically stratified water outside the borehole or if the packed interval straddles chemically heterogeneous zones.

The use of multilevel PDB samplers and other types of multilevel samplers (Ronen and others, 1987; Kaplan and others, 1991; Schirmer and others, 1995; Gefell and others, 1999; Jones and others, 1999) potentially can delineate some of the chemical stratification. Diffusion sampling and other sampling methodologies, however, can be influenced by vertical hydraulic gradients within the well screen or the sand pack. When vertical hydraulic gradients are present within the well, water contacting the PDB sampler may not be from a horizon adjacent to the PDB sampler. Rather, the water may represent a mixing of water from other contributing intervals within the borehole. In a screened well, even multilevel samplers with baffles to limit vertical flow in the well cannot prevent influences from

vertical flow in the gravel pack outside the well screen. Such vertical flow can result from small vertical differences in head with depth. A field test conducted by Church and Granato (1996) found that vertical head differences ranging from undetectable to 0.49 ft were sufficient to cause substantial flows (as much as 0.5 liters/minute) in the well bore.

QUALITY CONTROL AND ASSURANCE

The sources of variability and bias introduced during sample collection can affect the interpretation of the results. To reduce data variability caused during sampling, a series of quality-control samples should be utilized.

Replicate samples are important for the quality control of diffusion-sampler data. Sample replicates provide information needed to estimate the precision of concentration values determined from the combined sample-processing and analytical method and to evaluate the consistency of quantifying target VOCs. A replicate sample for water-filled diffusion samplers consists of two separate sets of VOC vials filled from the same diffusion sampler. Each set of VOC vials should be analyzed for comparison. Approximately 10 percent of the samplers should be replicated.

The length of the PDB sampler can be adjusted to accommodate the data-quality objectives for the sampling event. The length can be increased if additional volume is required for collection of replicate and matrix spike/matrix spike duplicate samples.

Trip blanks are used to determine whether external VOCs are contaminating the sample due to bottle handling and/or analytical processes not associated with field processing. Trip blanks are water-filled VOA vials prepared offsite, stored and transported with the other bottles used for collecting the environmental sample, and then submitted for analysis with the environmental sample. Consideration also should be given to the collection of a predeployment PDB trip blank to determine if the PDB samplers are exposed to extraneous VOCs prior to deployment. The predeployment trip blank should be a PDB sampler that is stored and transported with the field PDB samplers from the time of sampler construction to the time of deployment in the wells. An aliquot of the predeployment blank water should be collected from the PDB sampler in a VOA vial and submitted for analysis at the time of sampler deployment.

Water used to construct the diffusion samplers should be analyzed to determine the presence of background VOCs. Although many VOCs accidentally introduced into the diffusion-sampler water probably will reequilibrate with surrounding water once the diffusion samplers are deployed, some VOCs may become trapped within the diffusion-sampler water. For example, acetone, which is a common laboratory contaminant, does not easily move through the polyethylene diffusion samplers (Paul Hare, General Electric Company, oral commun., 1999). Thus, acetone inadvertently introduced into the diffusion-sample water during sampler construction may persist in the samplers, resulting in a false positive for acetone after sampler recovery and analysis.

SUMMARY

Water-filled passive diffusion bag (PDB) samplers described in this report are suitable for obtaining a variety of VOCs in ground water at monitoring wells. The suggested application for PDB samplers is for long-term monitoring of VOCs in ground-water wells at well-characterized sites. Where the screened interval is greater than 10 ft, the potential for contaminant stratification and/or intra-borehole flow within the screened interval is greater than in screened intervals shorter than 10 ft. It is suggested that the vertical distribution of contaminants be determined in wells having 10-ft-long well screens, and that both the vertical distribution of contaminants and the potential for intra-borehole flow be determined in wells having screens longer than 10 ft. A typical PDB sampler consists of a 1- to 2-ft-long low-density polyethylene lay-flat tube closed at both ends and containing deionized water. The sampler is positioned at the target horizon by attachment to a weighted line or fixed pipe.

The amount of time that the samplers should be left in the well prior to recovery depends on the time required by the PDB sampler to equilibrate with ambient water and the time required for environmental disturbances caused by sampler deployment to return to ambient conditions. The rate that water within the PDB sampler equilibrates with ambient water depends on multiple factors, including the type of compound being sampled and the water temperature. Concentrations of benzene, *cis*-1,2-dichloroethene, tetrachloroethene, trichloroethene, toluene, naphthalene, 1,2-dibromoethane, and total xylenes within the PDB samplers equilibrated with the concentrations in an

aqueous mixture of those compounds surrounding the samplers under laboratory conditions within approximately 48 hours at 21 °C. A subsequent laboratory study of mixed VOCs at 10 °C showed that tetrachloroethene and trichloroethene were equilibrated by about 52 hours, but other compounds required longer equilibration times. Chloroethane, *cis*-1,2-dichloroethene, *trans*-1,2-dichloroethene, and 1,1-dichloroethene were not equilibrated at 52 hours, but appeared to be equilibrated by the next sampling point at 93 hours. Vinyl chloride, 1,1,1-trichloroethane, 1,2-dichloroethane, and 1,1-dichloroethane were not equilibrated at 93 hours but were equilibrated by the next sampling point at 166 hours. Different equilibration times may exist for other compounds. Differences in equilibration times, if any, between single-solute or mixed-VOC solutions have not yet been thoroughly examined.

The samplers should be left in place long enough for the well water, contaminant distribution, and flow dynamics to restabilize following sampler deployment. Laboratory and field data suggest that 2 weeks of equilibration probably is adequate for many applications. Therefore, a minimum equilibration time of 2 weeks is suggested. In less permeable formations, longer equilibration times may be required. When deploying PDB samplers in waters colder than previously tested (10 °C) or for compounds without sufficient corroborating data, a side-by-side comparison with conventional methodology is advisable to justify the field equilibration time.

Following the initial equilibration period, the samplers maintain equilibrium concentrations with the ambient water until recovery. Thus, there is no specified maximum time for sampler recovery after initial equilibration. PDB samplers have routinely been left in ground waters having concentrations of greater than 500 ppm of TCE for 3 months at a time with no loss of bag integrity, and at one site, the PDB samplers were left in place in VOC-contaminated ground water for 1 year with no reported loss of sampler integrity. The effects of long-term (greater than 1 month) PDB-sampler deployment on sampler and sample integrity have not yet been thoroughly tested for a broad range of compounds and concentrations. In some environments, development of a biofilm on the polyethylene may be a consequence of long-term deployment. Investigations of semipermeable membrane devices

(SPMDs) have shown that the transfer of some compounds across a heavily biofouled polyethylene membrane may be reduced, but not stopped. If a heavy organic coating is observed on a PDB sampler, it is advisable to determine the integrity of the sample by comparing sampler results to a conventional sampling method concentrations before continuing to use PDB samplers for long-term deployment in that well.

PDB methodology is suitable for a broad variety of VOCs, including chlorinated aliphatic compounds and petroleum hydrocarbons. The samplers, however, are not suitable for inorganic ions and have a limited applicability for non-VOCs and for some VOCs. For example, although methyl-*tert*-butyl ether and acetone and most semivolatile compounds are transmitted through the polyethylene bag, laboratory tests have shown that the resulting concentrations were lower than in ambient water. The samplers should not be used to sample for phthalates because of the potential for the LDPE to contribute phthalates to the water sample.

When attempting to determine whether the use of PDB samplers is appropriate at a particular well, a common approach is to do a side-by-side comparison with a conventional sampling method. This approach is strongly suggested in wells having temporal concentration variability. In a well having relatively low temporal concentration variability, comparison of the PDB-sampler results to historical concentrations may provide enough information to determine whether the PDB samplers are appropriate for the well. In general, if the two approaches produce concentrations that agree within a range deemed acceptable by the local, state, and Federal regulatory agencies, then use of a PDB sampler in that well will provide VOC concentrations consistent with the historical record. If concentrations from the PDB sampler are higher than concentrations from the conventional method, then it is probable that the concentrations from the PDB sampler are an adequate representation of ambient conditions. If, however, the conventional method produces concentrations that are substantially higher than the concentrations found by using the PDB sampler, then the PDB sampler may or may not adequately represent local ambient conditions. In this case, the difference may be due to a variety of factors, including mixing or translocation due to hydraulic and chemical heterogeneity of the aquifer within the screened or open interval of the well and the relative permeability of the well screen.

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User's Guide for Polyethylene-Based Passive Diffusion Bag Samplers to Obtain Volatile Organic Compound Concentrations in Wells

Part 2: Field Tests

By Don A. Vroblesky, editor

U.S. Geological Survey

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Conversion Factors, Vertical Datum, Acronyms, and Abbreviations

Multiply	By	To obtain
<i>Length</i>		
inch (in.)	25.4	millimeter
foot (ft)	0.3048	meter
mile (mi)	1.609	kilometer
<i>Area</i>		
square mile (mi ²)	2.590	square kilometer
<i>Flow</i>		
foot per day (ft/d)	0.3048	meter per day
foot squared per day (ft ² /d)	0.09294	meter squared per day
gallons per minute (gal/min)	0.06308	liter per second
gallons per day (gal/d)	0.003785	cubic meter per day
inches per year (in/yr)	25.4	millimeters per year
<i>Volume</i>		
gallon (gal)	3.785	liter

Temperature is given in degrees Celsius (°C), which can be converted to degrees Fahrenheit (°F) by the following equation:
 $^{\circ}\text{F} = 9/5 (^{\circ}\text{C}) + 32$

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

Chemical concentration in water is expressed in metric units as milligrams per liter (mg/L) or micrograms per liter (µg/L).

Additional Abbreviations	
ft ³ /d	cubic feet per day
ft ³ /mg	cubic feet per milligram
g	gram
L	liter
µg	microgram
µm	micrometer
µL	microliter
mg	milligram
mL	milliliter
mL/min	milliliter per minute

User's Guide for Polyethylene-Based Passive Diffusion Bag Samplers to Obtain Volatile Organic Compound Concentrations in Wells

Part 2: Field Tests

By Don A. Vroblesky, *editor*

INTRODUCTION

This report presents six case studies where passive diffusion bag (PDB) samplers were tested under field conditions. The sites represent two U.S. Naval facilities [Naval Air Station (NAS) North Island, California; and Naval Industrial Reserve Ordnance Plant (NIROP) Fridley, Minnesota], and three U.S. Air Force facilities [Davis Global Communications, California; Hanscom Air Force Base (AFB), Massachusetts; and McClellan AFB, California]. The primary ground-water contaminants of interest were chlorinated hydrocarbons. Two independent studies included herein were done at McClellan AFB (Tunks and others, 2000; McClellan AFB Environmental Management Directorate, 2000). Because of the length of the McClellan AFB Environmental Management Directorate (2000) study, only a summarization of the report is included herein. The detailed report is available from McClellan AFB Environmental Management Directorate, 5050 Dudley Boulevard, Suite 3, McClellan AFB, California, 95652-1389.

Most of the case studies are previously published reports or summaries of previously published reports, some of which are authored by non-U.S. Geological Survey personnel. Therefore, the formatting of the individual reports varies, and not all formats are standard for the U.S. Geological Survey. Moreover, the methods used for these investigations preceded publication of standardized approaches for using PDB samplers in wells. Therefore, investigators should refer to Part 1 of this document for guidance on recommended methodology for PDB sampler applications, rather than to the case studies presented here.

PDB-sampler methodology was compared to conventional purging methods (purging at least three casing volumes) used at McClellan AFB and Davis Global Communications, and to low-flow methods used at NAS North Island and Hanscom AFB. Both conventional purging and low-flow purging were compared with using PDB samplers at NIROP Fridley. The study by Tunks and others at McClellan AFB compared the PDB samplers to conventional and low-flow techniques, as well as another type of diffusion device, the DMLS sampler.

The sites showed close correspondence between concentrations obtained by the PDB samplers and concentrations obtained by using other techniques at most tested locations. Most of the field studies also reported some disagreement between results from the PDB samplers and results from the comparative method at a few wells. The places where disagreements between results were observed are of interest because they illustrate differences between the sources of water for each type of sampling method. For example, in a well at Davis Global Communications where concentrations from the PDB samplers were lower than from the conventional purge, heat-pulse flowmeter testing was used to show that the water from the purged sampling probably was transported downward from a shallower contaminated aquifer during the well purge. When the well was not being pumped, however, the greatest amount of water entering the screen was from the sand layer adjacent to the screen. The data suggest that the PDB samplers provided concentrations characteristic of the aquifer under normal circumstances, whereas the pumped sample represented a mixture of water from the near vicinity of the well

screen, as well as contaminated water from a shallower horizon. Although the two methods did not agree, it appears that the PDB samplers provided results more characteristic of the aquifer adjacent to the screened interval.

Typically, other field studies also found that concentration differences between the PDB samplers and the pumping methodology used for comparison often could be attributed to an obscuring of the contaminant stratification by the mixing of water during pumping. Field evidence to support this hypothesis is shown in the reports on NAS North Island, Hanscom AFB, McClellan AFB, and NIROP Fridley. In general, the data show that even when the results of the PDB sampling and the conventional or low-flow-purging approaches disagree, the results of the PDB sampling often appear to accurately reflect the local concentrations, whereas those of the pumped sampling method reflect a mixing.

The investigation at McClellan AFB by Tunks and others, included in this report, shows a cost comparison for various sampling methods, however, some of these costs include a one-time investment for

the field test. Cost savings from more standard well-monitoring activities have been reported to range from 25 to 70 percent (Alexander and Lammons, 1999; Hare, 2000; U.S. Army Corps of Engineers, 2000; Brian Peters, OHM Remediation Services Corp., written commun., 2000). Cost savings of PDB sampling over conventional three-casing-purge sampling are described in the McClellan AFB Environmental Management Directorate report (2000), however, calculation errors obscure the actual amount of the savings.

Due to the availability of reports at the time of publication, the case studies included herein are limited to applications at sites where chlorinated aliphatic hydrocarbons are the primary contaminants. The case studies present data suggesting that PDB samplers can provide representative concentrations of the target compounds in a variety of environments. The method is a cost-effective, simple alternative to traditional sampling methodologies.

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Diffusion Sampler Evaluation of Chlorinated VOCs in Groundwater

By John Tunks, Peter Guest, and Javier Santillan

Diffusion Sampler Evaluation of Chlorinated VOCs in Groundwater

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ABSTRACT: Groundwater sample collection using diffusion samplers represents a relatively new technology that utilizes passive sampling methods for monitoring volatile organic compounds (VOCs) in groundwater. The potential benefits and cost savings of diffusion sampler use as an instrument for long-term monitoring are significant, as no purge waters are generated, and labor requirements for sampler installation and retrieval are minimal. The efficacy of diffusion samplers for evaluating chlorinated VOCs in groundwater was assessed. Using two types of diffusion samplers, groundwater samples were collected at discrete depths to assess vertical contamination profiles. Groundwater samples also were collected following low-flow/minimal drawdown purging and conventional purging techniques. Results obtained using the various sampling techniques suggest that the diffusion samplers provide comparable accuracy with and can be significantly less expensive than traditional sampling techniques.

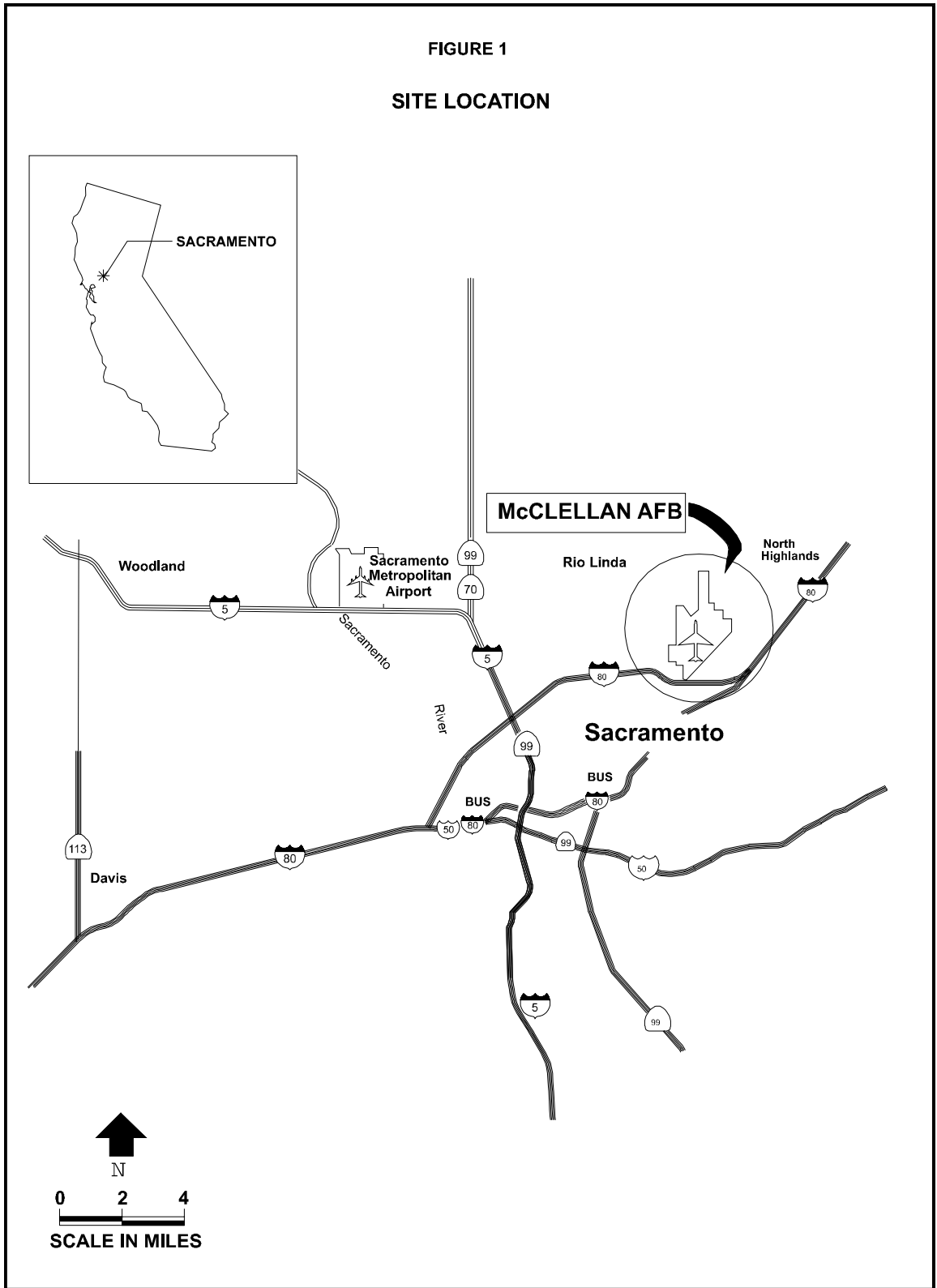
INTRODUCTION

Parsons Engineering Science, Inc. (Parsons ES) was retained by the US Air Force Center for Environmental Excellence, Technology Transfer Division (AFCEE/ERT) to perform an evaluation of passive groundwater diffusion sampling technology. The diffusion sampler evaluation is part of the AFCEE/ERT Remedial Process Optimization (RPO) demonstration project being performed at six Air Force bases (AFBs) nationwide. One of these bases, McClellan AFB, California (figure 1), was selected as the site for this evaluation. A field study was performed in August 1999 at a site on McClellan AFB where deep groundwater, more than 30 meters below ground surface, is contaminated with various chlorinated VOCs as a result of solvent disposal into burn pits during the 1940s through 1970s.

The objective of the diffusion sampler evaluation was to evaluate the efficacy of this groundwater sampling method relative to standard sampling methods. Field sampling was conducted using two types of diffusion samplers to collect groundwater samples from varying depths at selected monitoring wells. The diffusion samplers evaluated included the commercially available DMLS™ sampler (obtained from Johnson Screens, New Brighton, Minnesota in August 1999), and a sampler currently being developed and used by the US Geological Survey (USGS). The standard sampling methods used for comparison to the diffusion sampling results were:

1. Groundwater sampling following conventional purging of at least 3 casing-volumes of water and stabilization of water quality parameters (i.e., conventional sampling); and
2. Sampling following low-flow/minimal drawdown purging (i.e., micropurging). The groundwater samples were analyzed for total VOCs using US Environmental Protection Agency (USEPA) Method SW8260B/5030 (USEPA, 1994).

FIGURE 1
SITE LOCATION

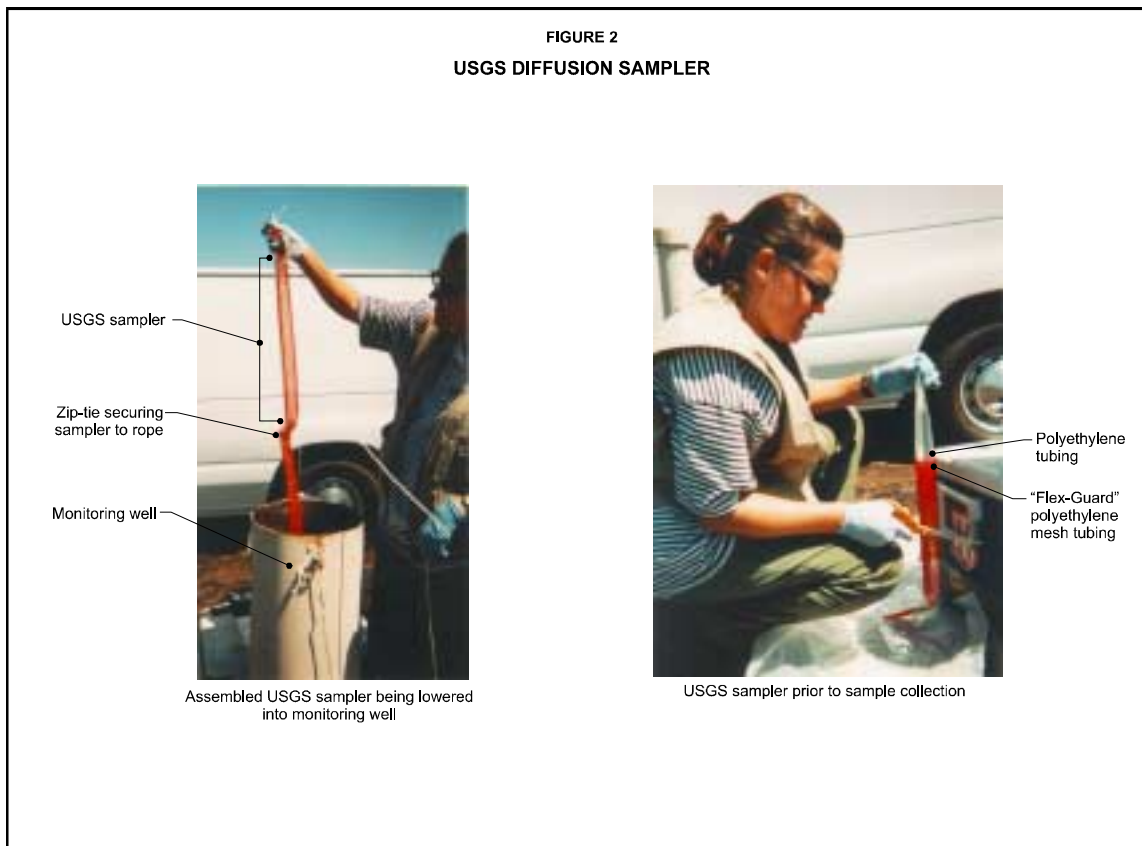


MATERIALS AND METHODS

Diffusion sampling is a relatively new technology designed to use passive sampling techniques that eliminate the need for well purging. A diffusive-membrane capsule is filled with deionized distilled water, sealed, mounted in a suspension device, and lowered to a specified depth in a monitoring well. Over time (no less than 72 hours), VOCs in the groundwater diffuse across the capsule membrane, and contaminant concentrations in the water inside the sampler attain equilibrium with the ambient groundwater. The sampler is subsequently removed from the well, and the water within the diffusion sampler is transferred to a sample container and submitted for analysis. The diffusive membranes evaluated in this study are rated for VOCs only. These membranes are not appropriate for monitoring larger or more electrically charged molecules.

Once a diffusion sampler is placed in a well, it remains undisturbed until equilibrium is achieved between the water in the well casing and the water in the diffusion sampler. Depending on the hydrogeologic characteristics of the aquifer, the diffusion samplers can reach equilibrium within 3 to 4 days (Vroblesky and Campbell, 1999); however for this evaluation, a minimum 14-day equilibrium period was used. Groundwater samples collected using the diffusion samplers are thought to be representative of water present within the well during the previous 24 to 72 hours.

USGS Sampler.—The standard USGS diffusion sampler, shown in figure 2, consists of water-filled, low-density polyethylene tubing, which acts as a semi-permeable membrane. The USGS sampler typically is constructed of a 45-centimeter (cm)-long section of 5.08-cm-diameter, 4-mil polyethylene tubing that is heat-sealed on both ends. The sampler holds approximately 300 milliliters (mL) of deionized distilled water. A longer 7.62-cm-diameter sampler that holds approximately 500 mL of water also is available if larger sample volumes are required. The sampler is placed in “flex-guard” polyethylene mesh tubing for abrasion protection, attached to a weighted rope, and lowered to a predetermined depth within the screened interval of a well. The rope is weighted to ensure that the sampling devices are positioned at the correct depth and that they do not float upward through the water column.



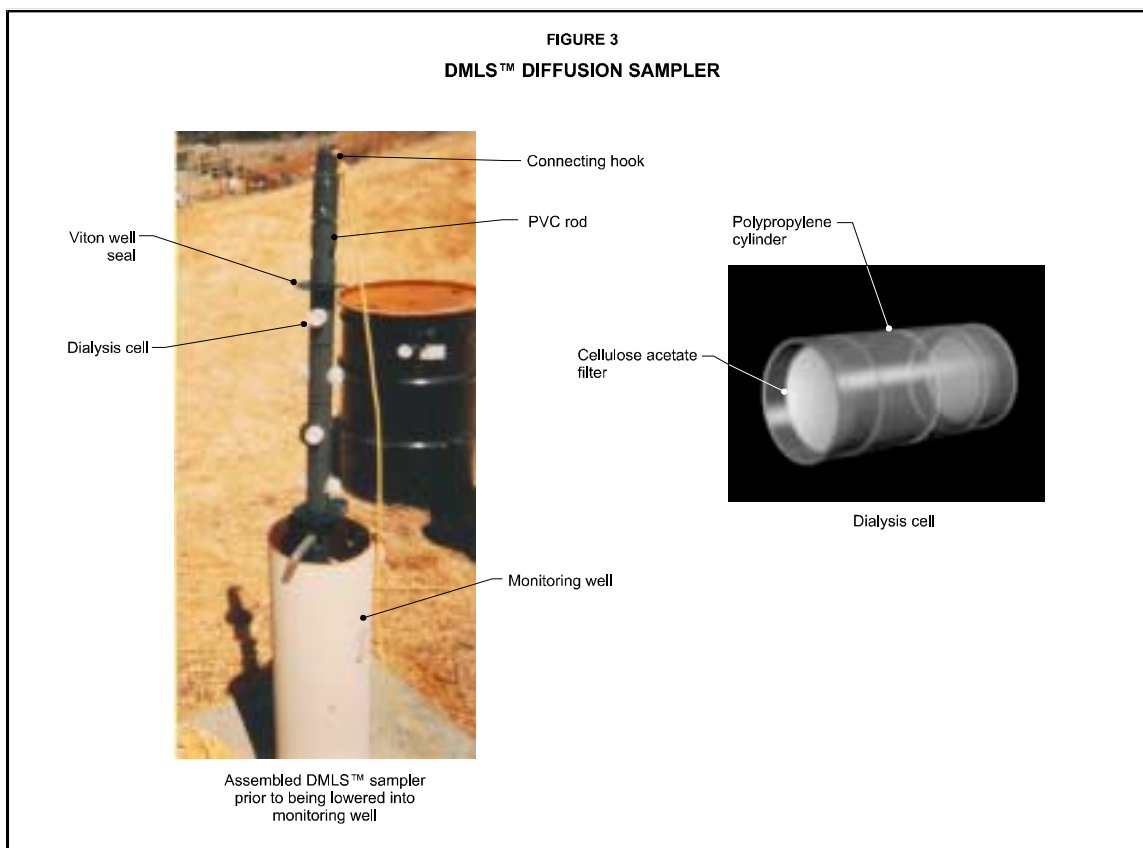
For this evaluation, multiple USGS samplers were placed end-to-end in three test monitoring wells to develop vertical contamination profiles. Upon recovery, the samplers were cut open, and water samples were transferred into 40-mL volatile organics analysis (VOA) vials. The samples were preserved and submitted for analysis.

DMLSTM Sampler.—The DMLS™ sampler, shown in figure 3, uses dialysis cells as passive collection devices. The dialysis cells are composed of a polypropylene cylinder that holds 38 mL of deionized distilled water. The cells have 0.2-micrometer cellulose acetate filters attached to each end of the cell that serve as the permeable membranes. The cells are mounted in cylindrical holes pre-drilled through a 152-cm-long polyvinyl chloride (PVC) rod, and are separated by viton spacers, or well seals, that fit the inner diameter of the well. The PVC rod can accommodate as many as 12 sampling cells (pre-drilled cylindrical hole spacing is 12.7 cm), and a string of up to 5 rods can be connected together for sampling over long screened well intervals.

Once loaded with the prepared dialysis cells, the PVC rods are lowered into a well to the desired depth within the screened interval, and are secured with a rope to the top of the well casing. A stainless steel weight is attached to the bottom of the deepest PVC rod to ensure that the samplers are positioned at the correct depth in the well, and that the PVC rods do not float through the water column.

Upon retrieval of the PVC rods, the dialysis cells are removed from the PVC rod, emptied into a decontaminated container for compositing, and then transferred to 40-mL VOA containers. The samples are preserved and sent to a laboratory for analysis.

Conventional Sampling.—Groundwater sampling using conventional well purging involves removing a large volume of water (3 to 5 well casing-volumes) from the well over a short time. The objective of conventional purging is to remove all water present within the well casing, as well as groundwater present in the surrounding well filter pack. Theoretically, by removing this water quickly, the “stagnant” water that resided in the well and filter



pack will be replaced with “fresh” groundwater from the surrounding formation with minimal mixing. The “fresh” groundwater that is then sampled is considered to be representative of the local groundwater. Rapid drawdown of the water level in a well is not uncommon, and often wells are purged dry using this method. Conventional purging is frequently performed using a bailer or a high-flow submersible pump (e.g., Grundfos Redi-Flo2 pump).

Micropurging.—The objective of micropurging is to remove a small volume of water at a low flow rate from a small portion of the screened interval of a well without mixing water among vertical zones. Ideally, by placing the inflow port of a pump at a prescribed depth within the screened interval of a well, and by withdrawing water at a slow rate, groundwater will be drawn from the aquifer into the well only in the immediate vicinity of the pump. This discrete-depth sampling allows for vertical definition of contamination in the aquifer. The pumping rate is adjusted to minimize drawdown. Because micropurging relies on a pumping rate that does not exceed the natural groundwater recharge rate, the water elevation in the well must be monitored to ensure that drawdown does not occur.

Field Activities.—Three monitoring wells were selected for use in this evaluation. In each of the wells, a maximum of three depth intervals spaced equally across the well screen were monitored using the different sampling methods. Using the two types of diffusion samplers as designed, it was necessary to perform the diffusion sampling consecutively, as samples from the two types of diffusion samplers could not be collected concurrently from the same interval within a well. To evaluate the potential changes in groundwater concentrations over the sampling periods (approximately 14 days between diffusion sampler collection events), conventional groundwater sampling was performed following completion of each diffusion sampling event. Significant differences in groundwater chemistry measured between the two sampling events could be normalized using the two sets of conventional groundwater data.

RESULTS AND DISCUSSION

Of the 67 analytes included in the SW8260B analysis, 17 were reported to have detectable concentrations in at least one of the samples submitted for analysis. For the purposes of comparing the analytical accuracy or comparability using the different sampling methods, only those analytes that were detected in at least 10 samples were considered in this study. These analytes include trichloroethene (TCE), *trans*-1,2-dichloroethene (DCE), *cis*-1,2-DCE, 1,1-DCE, 1,1-dichloroethane (DCA), 1,2-DCA, and 1,1,2-trichloroethane (TCA). A summary of analytical results for these analytes is presented in table 1.

The different methods of sample collection were evaluated using the following criteria: accuracy or comparability of data, other method-specific criteria, and cost. These criteria are described in the following sections.

Accuracy/Comparability of Data.—The analysis-of-variance (ANOVA) test was used to compare analytical data collected using the different sampling techniques. The limited number of samples available (as few as 3 per sampling method) precluded the use of linear statistical models in a quantitative manner. Therefore, the ANOVA was used in a qualitative manner to provide a “weight-of-evidence” support for data accuracy and similarity.

The ANOVA test returns a “p-value” between zero and one, indicating a “pass” or “fail” condition. A p-value of 0.05 or greater represents a pass, indicating that the distributions are similar at the 95-percent confidence level.

ANOVA is a parametric test, and it is common practice to verify that the data fit a parametric distribution prior to applying the tests. However, due to the limited number of samples in the data set, normality tests were not performed on the data sets before performing the ANOVA.

In instances where a nondetectable concentration of an analyte was reported for a sample, a value of zero was assigned for the purposes of the ANOVA testing only. For the conventional purging, each of the three depth intervals evaluated was assigned the same analytical value reported for the one sample collected from that well.

Table 1. Analytical results for samples

[µg/L, micrograms per liter]

Well ID	First Mobilization			Second Mobilization	
	USGS	Micropurge	Conventional	DMLS™	Conventional
TCE (µg/L)					
MW11	8 to 23	24	29	8 to 10	21
MW241	3.8 to 40	27 to 33	41	27 to 33	32
MW242	3.4 to 6	2.8 to 3.5	4	3.3 to 5.3	3.1
<i>trans</i>-1,2-DCE (µg/L)					
MW11	ND	ND	ND	ND	ND
MW241	ND to 1.2	0.90 to 0.98	1	0.77 to 1.4	0.99
MW242	ND	ND	ND	ND	ND
<i>cis</i>-1,2-DCE (µg/L)					
MW11	0.95 to 2.3	3.4	3.8	1.1 to 1.4	3.3
MW241	0.63 to 9.2	6.5 to 7.2	7.2	6 to 11	6.8
MW242	ND	ND	ND	ND	ND
1,1-DCE (µg/L)					
MW11	34 to 89	170	220	58 to 77	170
MW241	2.1 to 22	15 to 19	23	19 to 21	18
MW242	4.4 to 9	3.8 to 6.3	5.4	5.2 to 10	3.1
1,1-DCA (µg/L)					
MW11	0.66 to 1.6	1.6	1.7	0.54 to 0.69	1.5
MW241	0.36 to 4.4	3.5 to 3.6	3.6	2.9 to 4.3	3.4
MW242	ND	ND	ND	ND to 0.22	ND
1,1,2-TCA (µg/L)					
MW11	0.58 to 1.6	1.3	1.6	0.47 to 0.68	1.5
MW241	0.32	0.23 to 0.28	0.32	0.22 to 0.27	0.27
MW242	ND	ND	ND	ND	ND
1,2-DCA (µg/L)					
MW11	0.95 to 2.2	2.2	2	0.74 to 0.83	1.9
MW241	1.8 to 16	14 to 16	15	12 to 15	15
MW242	0.43 to 1.6	0.98 to 3.5	5.3	0.78 to 1.4	3.6

Notes:

"8 to 23" – Range of concentrations measured over sampled depth intervals.

ND – Not detected.

Data validation qualifiers did not affect the usability of the data for this evaluation and are therefore not included in table 1.

As presented in table 2, in all instances the p-values calculated for the populations of results for the different sampling methods exceeded 0.05. These ANOVA results indicate that there are no statistically significant differences among analytical results obtained using the four groundwater sampling techniques. Given that the evaluated diffusion samplers provide comparable accuracy with traditional sampling techniques, other criteria must be considered in evaluating the suitability of one sampling technique over another.

Table 2. ANOVA results

Analyte	p-value
1,1,2-TCA	0.74
1,1-DCA	0.99
1,1-DCE	0.47
1,2-DCA	0.88
cis-1,2-DCE	0.96
TCE	0.59
trans-1,2-DCE	0.99

Other Method-Specific Criteria.—Additional qualitative and semi-quantitative criteria were considered in this evaluation and are summarized in table 3.

Table 3. Summary of other method-specific criteria results

Criteria	USGS	DMLS™	Micropurge	Conventional
Ease of use	Excellent	Fair	Poor	Fair
Labor hours required per sample	0.66	1	2.75	3.66
Generation of IDW (liters)	< 1	< 1	100	500
Cost to provide dedicated equipment in each well	Low	High	Low	High
Decontamination required if dedicated equipment is not used	Minimal	High	Moderate	Moderate
Immediacy of sample availability	Slow	Slow	Rapid	Rapid
Can analytes other than VOCs be monitored?	No	No	Yes	Yes
Can vertical distribution of contaminants be evaluated?	Possible	Possible	Partial	No
Suitable for natural attenuation monitoring?	No	No	Yes	Partial

Supplemental to the criteria shown in table 3, concerns specific to the USGS and DMLS™ samplers were noted. Being placed in a well for potentially long periods, these samplers are susceptible to the effects of fluctuating groundwater elevations. If groundwater elevations decrease such that a portion of the diffusion sampler is exposed to air, the potential exists for volatilization of VOCs, which would compromise the samples collected from these devices.

A second concern was identified with the DMLS™ sampling device in that the sample volume of each dialysis cell is only 38 mL. When collecting samples for VOC analysis, the typical sample container is a 40 mL VOA, which will require more than one dialysis cell to fill.

As shown in table 3, many benefits can be realized through the use of diffusion samplers, however these devices also present limitations which may preclude their use in certain groundwater sampling applications.

Cost.—Cost estimates per sample for each of the four sampling methods evaluated are presented in table 4. The following expenses were considered in the development of a cost analysis for each different sampling method: labor, equipment, and disposal or management of investigation-derived waste (IDW). Some of the costs involved in these activities are one-time expenses that are not incurred each time a sample is collected (e.g., PVC rods for use with the DMLS™ samplers and stainless steel weights). Furthermore, labor and material costs can vary depending on the scope of the sampling event (e.g., it is less expensive on a unit-cost basis to collect 100 samples than to collect 5 samples). However, to present the most accurate estimate of costs associated with this evaluation, only the costs incurred during this field study were considered in the cost analysis. Labor costs were based on actual hours expended as documented in the field notes and the burdened labor rate for a typical field scientist. Equipment costs were taken directly from invoices (when available) or were estimated from vendor quotes. Costs associated with disposal or management of IDW can vary widely depending on the approach used. For this analysis, the only costs considered in the management of IDW are those dealing with containerizing the waste.

Table 4. Cost summary

Sampling technique	Cost per sample
USGS	\$65
DMLS™	\$555
Micropurge	\$308
Conventional	\$444

As noted, these costs are approximated based on the limited scope of this investigation. If these sampling technologies were applied to large-scale monitoring programs, a reduction in the per-sample cost would probably be realized due in part to reusable equipment that is associated with some of the sampling methods.

As shown in table 4, the cost per sample using the USGS diffusion sampler was substantially less than using any other methods. Conversely, the DMLS™ sampler per sample cost was substantially more than any other method.

CONCLUSIONS

The Air Force groundwater diffusion sampler evaluation indicates that diffusive sampling technology can be a cost-effective and accurate method for environmental groundwater monitoring of VOCs. However, use of diffusion samplers may not be appropriate for all applications. Of the diffusion sampling technologies evaluated, the USGS sampler is the recommended device based on the evaluation criteria presented herein. Additional comparisons between the different sampling technologies should be performed to develop a more robust data set upon which to base analytical result comparisons. Particularly, varying hydrogeologic settings (e.g., low-permeability to high-permeability aquifers) and increasing the number of wells in the evaluation would allow for more thorough evaluation of the comparability of the analytical data.

If natural attenuation monitoring is required, a combination of sampling techniques should be considered. For instance, annual monitoring of natural attenuation parameters can be performed using a traditional sampling method, while quarterly monitoring of VOCs can be accomplished using diffusion sampling technology.

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Diffusion Sampler Testing at Naval Air Station North Island, San Diego
County, California, November 1999 to January 2000
USGS Water-Resources Investigations Report 00-4182

By Don A. Vroblesky and Brian C. Peters

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ABSTRACT

Volatile organic compound concentrations in water from diffusion samplers were compared to concentrations in water obtained by low-flow purging at 15 observation wells at the Naval Air Station North Island, San Diego, California. Multiple diffusion samplers were installed in the wells. In general, comparisons using bladder pumps and diffusion samplers showed similar volatile organic carbon concentrations. In some wells, sharp concentration gradients were observed, such as an increase in *cis*-1,2-dichloroethene concentration from 100 to 2,600 micrograms per liter over a vertical distance of only 3.4 feet. In areas where such sharp gradients were observed, concentrations in water obtained by low-flow sampling at times reflected an average concentration over the area of influence; however, concentrations obtained by using the diffusion sampler seemed to represent the immediate vicinity of the sampler. When peristaltic pumps were used to collect ground-water samples by low-flow purging, the volatile organic compound concentrations commonly were lower than concentrations obtained by using diffusion samplers. This difference may be due to loss of volatiles by degassing under negative pressures in the sampling lines induced while using the peristaltic pump, mixing in the well screen, or possible short-circuiting of water from an adjacent depth. Diffusion samplers placed in buckets of free-phase jet fuel (JP-5) and Stoddard solvent from observation wells did not show evidence of structural integrity loss during the 2 months of

equilibration, and volatile organic compounds detected in the free-phase fuel also were detected in the water from the diffusion samplers.

INTRODUCTION

Low-density polyethylene diffusion samplers, filled with deionized water or air, have been shown to be an inexpensive alternative sampling method for volatile organic compounds (VOCs) in contaminated wells or in ground-water discharge zones beneath surface-water bodies (Vroblesky and others, 1996; Vroblesky and Robertson, 1996; Vroblesky and Hyde, 1997; Vroblesky and others, 1999; Gefell and others, 1999). The use of diffusion samplers in wells has generated substantial interest due to their capability to sample ground water without the need for prior well purging.

The Naval Air Station (NAS) North Island, in San Diego County, California, has been used since 1917 as an air station, harbor, and training base. The base is approximately 1 mile west across San Diego Bay from the San Diego metropolitan area (fig. 1). Activities at the base have resulted in ground-water contamination by a variety of compounds, including chlorinated aliphatic hydrocarbons and petroleum hydrocarbons. In addition, free-phase JP-5 jet fuel and Stoddard solvent (mineral spirits) are present locally in the ground water. Stoddard solvent is a refined petroleum product typically used as a thinning agent for paints, coatings, waxes, printing inks, and adhesives; a solvent in photocopy toners and in dry cleaning; and as a degreaser for engine parts.

The purpose of this report is to present the findings of an investigation to determine whether the use of polyethylene deionized-water-filled diffusion



Figure 1. Location of observation wells, Naval Air Station North Island, California.

samplers is a viable method of sampling VOCs in ground water at the base. VOC concentrations in water obtained from diffusion samplers set at multiple levels in wells are compared to VOC concentrations in water obtained from low-flow sampling. Diffusion samplers were placed in 15 observation wells, and 2 samplers were placed in buckets of free-phase JP-5 and Stoddard solvent.

Acknowledgments

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METHODS

Diffusion samplers were tested in 15 wells at NAS North Island, California. VOC concentrations in water from the diffusion samplers were compared to VOC concentrations in water from low-flow sampling ports open adjacent to each diffusion sampler. Low-flow sampling was accomplished by using a peristaltic pump at most sites and a bladder pump at selected sites.

Diffusion-Sampler Construction and Deployment

Each diffusion sampler consisted of a 2-inch-diameter, low-density polyethylene (LDPE) tube heat-sealed at both ends and containing deionized water. On the outside of each sampler, an LDPE mesh provided abrasion protection. This sampling methodology is patented (patent number 5,804,743) and is available for non-exclusive licensing from the U.S. Geological Survey Technology Enterprise Office, Mail Stop 211, National Center, 12201 Sunrise Valley Drive, Reston, Virginia (telephone 703 648-4450; fax 703 648-4408).

Diffusion samplers were attached to intakes of bladder pumps by means of plastic cable ties. Attached to each remaining diffusion sampler was a Tygon tube extending from the sampler to land surface. The tubing was secured to the diffusion sampler and to a weighted line at approximately 10-foot intervals by using plastic cable ties. The purpose of the tubing was to allow ground water to be collected adjacent to each diffusion sampler by using low-flow methodology with a peristaltic pump.

The diffusion samplers were deployed in 15 wells at NAS North Island during November 11, 1999 (table 1). All wells were constructed of 4-inch-diameter casing. The samplers were attached by plastic cable ties to either a weighted line or a 1/2-inch (outside diameter) PVC pipe. When multiple sections of PVC pipe were required to reach the top of the casing, the sections were joined using stainless-steel screws. The PVC pipe was secured to the top of the well casing to prevent the diffusion samplers from shifting during the equilibration period.

Two of the sampled wells (PW-15 and PW-55) contained floating nonaqueous-phase liquid (LNAPL) consisting of free-phase petroleum and Stoddard solvent. To install diffusion samplers in these wells, a

Table 1. Summary of well information, Naval Air Station North Island, California

[ft, feet; ft bls, feet below land surface; ft msl, feet relative to mean sea level; Elev., elevation; NM, not measured; NA, not available; TOC, top of casing; A, bladder pump attached to the diffusion sampler; B, peristaltic pump using tubing attached to individual diffusion samplers; C, same as B, except one depth was sampled using a bladder pump attached to a diffusion sampler; D, same as B, except the well was resampled using a bladder pump following removal of the diffusion samplers]

Site or building designation	Well identifier	Depth to screen top(ft bls)	Depth to screen bottom (ft bls)	Saturated screen length (ft)	Depth to water (ft bls)	Elev. of water (ft msl)	Low-flow sampling method
653	MW-10	5	20.0	13.0	7.01	2.65	B
653	MW-13A	4	14.0	8.18	6.01	1.81	D
653	MW-13B	24.3	29.2	5.00	6.15	1.53	D
653	MW-13C	44.8	49.8	5.00	6.00	1.61	D
472	MW-68 C2	37	63.0	25.0	NM	NA	B
472	MW-68A	14	24.0	2.76	21.38	2.34	B
472	MW-68B	33	40.0	5.00	21.42	2.33	B
472	MW-68C	64.3	70.5	5.00	21.6	1.99	B
379	PW-15	20	35.0	9.94	23.34	2.61	B
379	PW-55	20	35.0	9.33	24.32	2.34	B
379	PW-66	20	35.0	10.0	25.10	2.40	B
Site 11	MW-12	30	39.7	13.7	NM	NA	C
Site 11	MW-5D	NA	60.0	35.5	NM	NA	C
Site 11	MW-9	23	31.9	4.10	28.18	5.64	A
Site 2	S2-MW-6A	5	20.0	14.3	5.64	2.35	B

rubber cap was placed on the lower end of a section of 2-inch-diameter PVC pipe and lowered into the well to a depth below the LNAPL. The top end of the pipe extended to land surface. A smaller diameter pipe then was used to pound out the rubber cap, which was recovered from the well along the outside of the 2-inch-diameter pipe by means of a rope attached to the cap. The diffusion samplers were lowered into the well through the 2-inch-diameter pipe, thereby avoiding direct contact with the LNAPL. The pipe was secured in place to allow the diffusion sampler to be recovered without contact with the LNAPL.

Collection of Pumped Ground-Water Samples

The diffusion samplers were allowed to remain undisturbed in the well water for 65 to 71 days (table 2). The wells were sampled at the time of sampler recovery using low-flow techniques. Low-flow sampling consisted of purging the well by means of using a dedicated bladder pump or a peristaltic pump connected to the Tygon tubing that had been attached to each of the diffusion samplers prior to deployment. Purging was done at a rate of 120 milliliters per minute until measurements of pH, water temperature, and specific conductance stabilized. In general, purging involved about 20 minutes of pumping and removal of less than 1 gallon of water from each sampling port. Decontamination of equipment was not required because each sampling interval had dedicated tubing.

A variety of methods were used to retrieve the diffusion samplers and to low-flow sample the well. The first method of sample retrieval involved recovering the diffusion sampler from the well immediately following low-flow sampling by using a bladder pump from the depth at which the diffusion sampler had equilibrated (well MW-9 only). A second method involved using a peristaltic pump to low-flow sample ground water adjacent to each of the diffusion samplers by means of the dedicated Tygon tubing attached to each diffusion sampler. The depths were low-flow sampled beginning with the shallowest and proceeding to the deepest. In some wells (MW-5 and MW-12), one of the depths was sampled using a dedicated bladder pump while the remaining depths were sampled using a bladder pump attached to dedicated tubing. Two wells (MW-13A and MW-13B) were low-flow sampled by using a peristaltic pump, the diffusion samplers were recovered, a bladder pump was inserted into each well, and the wells were then immediately

resampled by low-flow methodology using the bladder pump.

The diffusion samplers were recovered from the wells by means of the attached weighted line or PVC pipe. The samplers were cut open, and the water was slowly decanted into glass vials pretreated with hydrochloric acid. The water samples were sent to a contract laboratory for analysis by using Environmental Protection Agency Method 8260B (U.S. Environmental Protection Agency, 1999). Replicate samples were collected from approximately 10 percent of the sampling sites. In general, both diffusion samples and low-flow samples compared well with their respective replicate samples (tables 3 and 4).

RESULTS AND DISCUSSION

VOC concentrations in water obtained from diffusion samplers were similar to concentrations obtained by using low-flow sampling methods for most of the tested wells (tables 5 and 6, respectively). As will be shown, most concentration differences between the two sampling methods probably can be attributed to VOC degassing during peristaltic-pump sampling or to in-well mixing.

Comparison of Diffusion-Sampler Results to Bladder-Pump Results

Tests showing the most direct comparison between diffusion sampling and low-flow sampling were in wells where a bladder pump was used to low-flow sample. The test producing the least amount of well-water disturbance was in well MW-9 where a diffusion sampler was recovered immediately following low-flow sampling using a bladder pump from the same depth. Concentrations of 1,1-dichloroethene (1,1-DCE) and trichloroethene (TCE) obtained using the diffusion sampler agreed well (12 and 3 percent difference, respectively) with those obtained using the bladder pump (table 7). The difference is about the same as the differences (approximately 12 percent) in 1,1-DCE and TCE concentrations measured in replicate samples collected by using a dedicated bladder pump at well MW-5D (table 4). Thus, 12 percent is within the sample-collection variability for 1,1-DCE and TCE. Agreement between the methods was poorer for tetrachloroethene (PCE) concentrations. The PCE concentration in water from the diffusion sampler was 21 percent lower than the concentration in water

Table 2. Sampler deployment and recovery information, Naval Air Station North Island, California, November 1999 to January 2000

[repl, replicate sample; NA, not applicable; *, low-flow bladder-pump sample; #, data from OHM Remediation Services Corporation (2000)]

Site or building designation	Well identifier	Sampling interval identifier	Low-flow sample laboratory identifier	Diffusion-sampler laboratory identifier	Depth to diffusion-sampler center (ft bls)	Date installed	Date recovered	Number of days diffusion samplers were in wells
653	MW-10	A	779679-0091	779679-0099	7.75	11/12/99	1/18/00	67
653	MW-10	B	779679-0092	779679-0100	9.15	11/12/99	1/18/00	67
653	MW-10	C	779679-0093	779679-0101	11.1	11/12/99	1/18/00	67
653	MW-10	D	779679-0094	779679-0102	13.1	11/12/99	1/18/00	67
653	MW-10	E	779679-0095	779679-0103	15.1	11/12/99	1/18/00	67
653	MW-10	F	779679-0096	779679-0104	17.1	11/12/99	1/18/00	67
653	MW-10	G	779679-0097	779679-0105	18.8	11/12/99	1/18/00	67
653	MW-10	G repl	779679-0098	NA	18.8	11/12/99	1/18/00	67
653	MW-13A	A	779679-0030	779679-0042	6.50	11/10/99	1/17/00	68
653	MW-13A	B	779679-0031	779679-0043	7.95	11/10/99	1/17/00	68
653	MW-13A	C	779679-0032	779679-0044	9.35	11/10/99	1/17/00	68
653	MW-13A	D	779679-0033	779679-0045	10.9	11/10/99	1/17/00	68
653	MW-13A	E	779679-0034	779679-0046	12.4	11/10/99	1/17/00	68
653	MW-13A*#	NA	NA	NA	NA	NA	1/17/00	NA
653	MW-13B	A	779679-0035	779679-0047	24.8	11/10/99	1/17/00	68
653	MW-13B	B	779679-0036	779679-0048	26.1	11/10/99	1/17/00	68
653	MW-13B	C	779679-0037	779679-0049	27.5	11/10/99	1/17/00	68
653	MW-13B*#	NA	NA	NA	NA	NA	1/17/00	NA
653	MW-13C	A	779679-0038	779679-0054	45.4	11/10/99	1/17/00	68
653	MW-13C	B	779679-0039	779679-0055	46.6	11/10/99	1/17/00	68
653	MW-13C	B repl	779679-0041	NA	46.6	11/10/99	1/17/00	68
653	MW-13C	C	779679-0040	779679-0056	48.0	11/10/99	1/17/00	68
653	MW-13C*#	NA	NA	NA	NA	NA	1/17/00	NA
472	MW-68A	A	779679-0023	779679-0025	21.7	11/9/99	1/17/00	69
472	MW-68A	B	779679-0024	779679-0026	23.0	11/9/99	1/17/00	69
472	MW-68B	A	779679-0016	779679-0020	34.5	11/9/99	1/17/00	69
472	MW-68B	B	779679-0017	779679-0021	37.0	11/9/99	1/17/00	69
472	MW-68B	B repl	779679-0018	NA	37.0	11/9/99	1/17/00	69
472	MW-68B	C	779679-0019	779679-0022	38.5	11/9/99	1/17/00	69
472	MW-68C	A	779679-0027	779679-0050	56.0	11/9/99	1/17/00	69
472	MW-68C	B	779679-0028	779679-0051	57.5	11/9/99	1/17/00	69
472	MW-68C	C	779679-0117	779679-0116	59.0	11/9/99	1/19/00	71

Table 2. Sampler deployment and recovery information, Naval Air Station North Island, California, November 1999 to January 2000—Continued

[repl, replicate sample; NA, not applicable; *, low-flow bladder-pump sample; #, data from OHM Remediation Services Corporation (2000)]

Site or building designation	Well identifier	Sampling interval identifier	Low-flow sample laboratory identifier	Diffusion-sampler laboratory identifier	Depth to diffusion-sampler center (ft bls)	Date installed	Date recovered	Number of days diffusion samplers were in wells
472	MW-68C2	A	779679-0166	779679-0181	37.2	11/11/99	1/20/00	70
472	MW-68C2	B	779679-0169	779679-0182	39.1	11/11/99	1/20/00	70
472	MW-68C2	B repl	NA	779679-0183	39.1	11/11/99	1/20/00	70
472	MW-68C2	C	779679-0171	779679-0184	40.5	11/11/99	1/20/00	70
472	MW-68C2	D	779679-0173	779679-0185	42.0	11/11/99	1/20/00	70
472	MW-68C2	E	779679-0176	779679-0186	44.1	11/11/99	1/20/00	70
472	MW-68C2	F	779679-0178	779679-0187	46.0	11/11/99	1/20/00	70
472	MW-68C2	G	779679-0180	779679-0188	47.8	11/11/99	1/20/00	70
472	MW-68C2	H	779679-0179	779679-0189	49.8	11/11/99	1/20/00	70
472	MW-68C2	I	779679-0177	779679-0190	51.9	11/11/99	1/20/00	70
472	MW-68C2	J	779679-0174	779679-0191	53.9	11/11/99	1/20/00	70
472	MW-68C2	J repl	779679-0175	NA	53.9	11/11/99	1/20/00	70
472	MW-68C2	K	779679-0172	779679-0192	55.5	11/11/99	1/20/00	70
472	MW-68C2	L	779679-0170	779679-0193	57.5	11/11/99	1/20/00	70
472	MW-68C2	M	779679-0168	779679-0194	59.5	11/11/99	1/20/00	70
472	MW-68C2	M repl	NA	779679-0195	59.5	11/11/99	1/20/00	70
472	MW-68C2	N	779679-0167	779679-0196	61.5	11/11/99	1/20/00	70
379	PW-15	A	779679-0083	779679-0089	25.4	11/13/99	1/19/00	67
379	PW-15	B	779679-0084	779679-0118	27.1	11/13/99	1/19/00	67
379	PW-15	B repl	NA	779679-0119	27.1	11/13/99	1/19/00	67
379	PW-15	C	779679-0085	779679-0156	28.5	11/13/99	1/19/00	67
379	PW-15	D	779679-0086	779679-0157	30.2	11/13/99	1/19/00	67
379	PW-15	E	779679-0087	779679-0158	31.7	11/13/99	1/19/00	67
379	PW-15	E repl	NA	779679-0159	31.7	11/13/99	1/19/00	67
379	PW-15	F	779679-0088	779679-0164	33.1	11/13/99	1/19/00	67
379	PW-55	A	779679-0077	779679-0109	27.1	11/13/99	1/19/00	67
379	PW-55	B	779679-0078	779679-0110	28.9	11/13/99	1/19/00	67
379	PW-55	B repl	NA	779679-0114	28.9	11/13/99	1/19/00	67
379	PW-55	C	779679-0079	779679-0111	30.6	11/13/99	1/19/00	67
379	PW-55	D	779679-0080	779679-0112	31.9	11/13/99	1/19/00	67
379	PW-55	E	779679-0081	779679-0113	33.1	11/13/99	1/19/00	67
379	PW-55	E repl	779679-0082	779679-0115	33.1	11/13/99	1/19/00	67
379	PW-66	A	779679-0106	779679-0145	25.5	11/10/99	1/18/00	69
379	PW-66	B	779679-0107	779679-0146	27.3	11/10/99	1/18/00	69
379	PW-66	B repl	NA	779679-0151	27.3	11/10/99	1/18/00	69

Table 2. Sampler deployment and recovery information, Naval Air Station North Island, California, November 1999 to January 2000—Continued

[repl, replicate sample; NA, not applicable; *, low-flow bladder-pump sample; #, data from OHM Remediation Services Corporation (2000)]

Site or building designation	Well identifier	Sampling interval identifier	Low-flow sample laboratory identifier	Diffusion-sampler laboratory identifier	Depth to diffusion-sampler center (ft bls)	Date installed	Date recovered	Number of days diffusion samplers were in wells
379	PW-66	C	779679-0073	779679-0147	29.1	11/10/99	1/18/00	69
379	PW-66	D	779679-0108	779679-0148	30.8	11/10/99	1/18/00	69
379	PW-66	E	779679-0074	779679-0149	32.3	11/10/99	1/18/00	69
379	PW-66	F	779679-0075	779679-0150	33.9	11/10/99	1/18/00	69
379	PW-66	F repl	NA	779679-0152	33.9	11/10/99	1/18/00	69
Site 11	MW-12	A	779679-0006	779679-0012	30.5	11/13/99	1/17/00	65
Site 11	MW-12	B	779679-0007	779679-0013	32.1	11/13/99	1/17/00	65
Site 11	MW-12	C	779679-0008	779679-0014	33.7	11/13/99	1/17/00	65
Site 11	MW-12*	D	779679-0009	779679-0057	35.1	11/13/99	1/17/00	65
Site 11	MW-12*	D repl	NA	779679-0060	35.1	11/13/99	1/17/00	65
Site 11	MW-12	E	779679-0010	779679-0058	36.9	11/13/99	1/17/00	65
Site 11	MW-12	F	779679-0011	779679-0059	38.5	11/13/99	1/17/00	65
Site 11	MW-5D	A	779679-0121	779679-0128	50.8	11/12/99	1/18/00	67
Site 11	MW-5D	B	779679-0122	779679-0129	52.3	11/12/99	1/18/00	67
Site 11	MW-5D	C	779679-0123	779679-0130	54.2	11/12/99	1/18/00	67
Site 11	MW-5D	C repl	NA	779679-0134	54.2	11/12/99	1/18/00	67
Site 11	MW-5D*	D	779679-0124	779679-0131	55.75	11/12/99	1/18/00	67
Site 11	MW-5D	D repl	779679-0125	NA	55.75	11/12/99	1/18/00	67
Site 11	MW-5D	E	779679-0126	779679-0132	57.4	11/12/99	1/18/00	67
Site 11	MW-5D	F	779679-0127	779679-0133	59.0	11/12/99	1/18/00	67
Site 11	MW9*	NA	779679-0154	779679-0155	31	11/12/99	1/19/00	68
Site 2	S2-MW6A	A	779679-0062	779679-0135	6.5	11/13/99	1/18/00	66
Site 2	S2-MW6A	B	779679-0063	779679-0136	7.85	11/13/99	1/18/00	66
Site 2	S2-MW6A	C	779679-0064	779679-0137	9.2	11/13/99	1/18/00	66
Site 2	S2-MW6A	D	779679-0065	779679-0138	10.6	11/13/99	1/18/00	66
Site 2	S2-MW6A	E	779679-0066	779679-0139	11.95	11/13/99	1/18/00	66
Site 2	S2-MW6A	F	779679-0067	779679-0140	13.3	11/13/99	1/18/00	66
Site 2	S2-MW6A	G	779679-0068	779679-0141	14.65	11/13/99	1/18/00	66
Site 2	S2-MW6A	H	779679-0069	779679-0142	16.05	11/13/99	1/18/00	66
Site 2	S2-MW6A	I	779679-0070	779679-0143	17.5	11/13/99	1/18/00	66
Site 2	S2-MW6A	J	779679-0071	779679-0144	18.95	11/13/99	1/18/00	66
Site 2	S2-MW6A	J repl	779679-0072	NA	18.95	11/13/99	1/18/00	66

Table 3. Comparison of replicate samples collected by diffusion-sampler methodology, Naval Air Station North Island, California, January 2000

[repl, replicate sample; ft bls, feet below land surface; (µg/L, micrograms per liter; J, estimated value; U, value was below the analytical quantitation limit; 11DCA, 1,1-dichloroethane; 11DCE, 1,1-dichloroethene; *c*DCE, *cis*-1,2-dichloroethene; TCE, trichloroethene]

Well identifier and (depth code)	Depth to diffusion sampler center (ft bls)	Site or building designation	11DCA (µg/L)	11DCE (µg/L)	<i>c</i> DCE (µg/L)	Ethylbenzene (µg/L)	TCE (µg/L)	Vinyl chloride (µg/L)	Total xylenes (µg/L)
MW-12 (D)	35.1	Site 11	86 J	1,500	100	100 U	1,800	100 U	100 U
MW-12 (D-repl)	35.1	Site 11	89 J	1,500	110	100 U	1,700	100 U	100 U
MW-5D (C)	54.2	Site 11	170	2,800 E	61	50 U	930	50 U	50 U
MW-5D (C repl)	54.2	Site 11	170	2,900 E	61	50 U	930	50 U	50 U
MW-68C2 (B)	39.1	472	2,500 U	4,100	1,000 J	2,500 U	47,000	2,500 U	2,500 U
MW-68C2 (B repl)	39.1	472	5,000 U	4,300 J	960 J	5,000 U	52,000	5,000 U	5,000 U
MW-68C2 (M)	59.5	472	500 U	350 J	500 U	500 U	7,000	500 U	500 U
MW-68C2 (M repl)	59.5	472	500 U	360 J	500 U	500 U	6,800	500 U	500 U
PW-15 (B)	27.1	379	52	8	130	15	7	72	52
PW-15 (B repl)	27.1	379	54	8	130	17	5 J	75	57
PW-15 (E)	31.7	379	500 U	500 U	1,900	500 U	5,500	500 U	500 U
PW-15 (E repl)	31.7	379	500 U	500 U	1,900	500 U	5,600	500 U	500 U
PW-55 (B)	28.9	379	2,500 U	2,500 U	6,500	2,500 U	39,000	2,500 U	2,500 U
PW-55 (B repl)	28.9	379	2,500 U	2,500 U	6,700	2,500 U	36,000	2,500 U	2,500 U
PW-55 (E)	33.1	379	2,500 U	2,500 U	6,300	2,500 U	33,000	2,500 U	2,500 U
PW-55 (E repl)	33.1	379	2,500 U	2,500 U	6,100	2,500 U	31,000	2,500 U	2,500 U
PW-66 (B)	28.9	379	500 U	500 U	3,400	500 U	9,000	500 U	500 U
PW-66 (B repl)	28.9	379	500 U	500 U	3,200	500 U	9,200	500 U	500 U
PW-66 (F)	33.1	379	5 U	130	5 U	5 U	13	5 U	5 U
PW-66 (F repl)	33.1	379	5 U	120	5 U	5 U	18	5 U	5 U

obtained using the bladder pump (table 7). The reason for the difference in tetrachloroethene concentrations is not known.

The data from well MW-5 (site 11) show that the diffusion samplers performed favorably. At well MW-5, where a bladder pump was used to obtain water adjacent to a diffusion sampler and where peristaltic pumps were used at the other depths, the difference between the TCE concentration in water from the adjacent diffusion sampler and the average concentration (300 µg/L) in water from the bladder pump was relatively small (17-percent difference) (table 7). Moreover, the higher TCE concentration in water from the diffusion sampler compared to the concentration in water from the bladder pump implies that the sample collected by the diffusion method was more discrete than the sample collected by using the bladder pump.

A comparison between diffusion samples and a bladder pump sample at well MW-12 showed that the TCE concentration in water from the diffusion sampler was similar to the TCE concentration in water from the bladder pump (1,800 and 2,100 µg/L, respectively); however, the *cis*-1,2-dichloroethene (*c*DCE) concentration in water from the diffusion sampler was substantially lower (78 percent) than the concentration in water from the bladder pump (table 7 and fig. 2). This difference may be due to in-well mixing by low-flow sampling in a chemically stratified part of the screened interval. Data from diffusion samplers show that the VOC concentrations substantially increased with depth over a distance of only 3.4 ft and that the bladder pump was positioned at a transition zone between two depths of differing concentrations (table 5 and fig. 2). The bladder pump was sampled

Table 4. Comparison of replicate samples collected by low-flow methodology, Naval Air Station North Island, California, January 2000

[repl, replicate sample; *, sample collected by using bladder pump - low-flow samples without * were collected by using a peristaltic pump; #, data from OHM Remediation Services Corp. (2000); ft bls, feet below land surface; (µg/L, micrograms per liter; J, estimated value; U, value was below the analytical quantitation limit; 11DCA, 1,1-dichloroethane; 11DCE, 1,1-dichloroethene; cDCE, *cis*-1,2 dichloroethene; TCE, trichloroethene]

Well Identifier and (depth code)	Depth to diffusion sampler center (ft bls)	Site or building designation	11DCA (µg/L)	11DCE (µg/L)	cDCE (µg/L)	Ethylbenzene (µg/L)	TCE (µg/L)	Vinyl chloride (µg/L)	Total xylenes (µg/L)
MW-10 (G)	18.8	653	5 U	0 J	5 U	5 U	6	5 U	3 J
MW-10 (G repl)	18.8	653	5 U	5 U	1 J	5 U	9	5 U	5 U
MW-13B**	26	653	5 U	5 U	3,100	5 U	5 U	1,600	5 U
MW-13B** (repl)	26	653	5 U	5 U	3,200	5 U	5 U	1,400	5 U
MW-13C (B)	46.6	653	5 U	5 U	3 J	5 U	5 U	5 U	5 U
MW-13C (B repl)	46.6	653	5 U	5 U	2 J	5 U	5 U	5 U	5 U
MW-5D (D)*	55.7	Site 11	51	760	23 J	25 U	320	25 U	25 U
MW-5D (D repl)*	55.7	Site 11	44	670	22 J	25 U	280	25 U	25 U
MW-68C2 (J)	53.9	472	2,500 U	2,500 J	2,500 U	2,500 U	38,000	2,500 U	2,500 U
MW-68C2 (J repl)	53.9	472	2,500 U	2,600	2,500 U	2,500 U	38,000	2,500 U	2,500 U
MW-68B (B)	37.0	472	5,000 U	4,400 J	5,000 U	5,000 U	34,000	5,000 U	5,000 U
MW-68B (B repl)	37.0	472	5,000 U	4,900 J	5,000 U	5,000 U	33,000	5,000 U	5,000 U
PW-55 (E)	33.1	379	2,500 U	2,500 U	5,500	2,500 U	29,000	2,500 U	2,500 U
PW-55 (E repl)	33.1	379	2,500 U	2,500 U	5,700	2,500 U	29,000	2,500 U	2,500 U
S2-MW-6A (J)	18.9	Site 2	5 U	5 U	5 U	5 U	5 U	5 U	5 U
S2-MW-6A (J repl)	18.9	Site 2	5 U	5 U	5 U	5 U	5 U	5 U	5 U

following low-flow sampling from three overlying depths using a peristaltic pump; thus, the concentration interface potentially shifted upward toward the bladder pump intake. It is possible that in-well mixing was more pronounced for cDCE than for TCE because there was a greater percentage of change in concentrations with depth for cDCE than for TCE. The cDCE concentration increased by a factor of 26 (100 to 2,600 µg/L) over a depth of 3.4 feet, whereas TCE increased by only a factor of 4.6 over the same depth interval (1,700 to 7,800 µg/L) (fig. 2). The VOC concentration data indicate that in well MW-12, the diffusion samplers collected point samples of ground water, whereas the bladder pump either collected water from a greater radius of influence or from water induced up the well bore by low-flow sampling at shallower depths.

Wells MW-13A and MW-13B were tested using diffusion samplers and low-flow sampling with a peristaltic pump. Following sample collection with the peristaltic pump, the diffusion samplers were recovered and the wells were sampled by using a bladder pump (bladder-pump data from OHM Remediation Services Corporation, 2000). The data show that at well MW-13A, the cDCE and vinyl chloride concentrations in water obtained using the bladder pump were within the concentration ranges for water obtained from diffusion samplers that bracketed the depth interval of the bladder pump intake (table 7). At well MW-13B, the cDCE concentration also was within the range measured in those diffusion samplers bracketing the depth of the bladder pump intake (table 7 and fig. 3). Although vinyl chloride concentrations differed between the two methods by 16 to

Table 5. Concentrations of selected chlorinated volatile organic compounds in water from diffusion and low-flow sampling, Naval Air Station North Island, California, January 2000

[µg/L, micrograms per liter; D, sample was diluted; U, value was below the analytical quantitation limit; J, estimated value; NA, not applicable; E, the detected result is between the sample-specific estimated quantitation and the method detection limit; *, sample collected by using bladder pump; #, data from OHM Remediation Services Corporation (2000)]

Well	Depth (feet)	1,1-Dichloroethane (µg/L)		1,1-Dichloroethene (µg/L)		<i>cis</i> -1,2-Dichloroethene (µg/L)		Tetrachloroethene (µg/L)		Trichloroethene (µg/L)		Vinyl chloride (µg/L)	
		Diffusion	Low flow	Diffusion	Low flow	Diffusion	Low flow	Diffusion	Low flow	Diffusion	Low flow	Diffusion	Low flow
MW-5D	50.8	260	200	1,500	1,900 D	66	62	50 U	25 U	690	660	50 U	25 U
MW-5D	52.3	260	210	3,600	4,300 D	120 J	63	250 U	50 U	1,200	1,100	250 U	50 U
MW-5D	54.2	170	95 J	3,200 D	1,400	61	36 J	50 U	100 U	930	590	50 U	100 U
MW-5D*	55.8	65	51	1,500 D	760	27	23 J	5 U	25 U	510 D	320	5 U	25 U
MW-5D	57.4	50 U	20	460	360 D	50 U	11	50 U	5 U	160	120	50 U	5 U
MW-5D	59.0	50 U	6	190	94	10 J	6	50 U	5 U	55	36	50 U	5 U
MW-9*	27.6	270	270	4,000	3,500	250 U	250 U	260	330	3,300	3,200	250 U	250 U
MW-10	7.8	5 U	5 U	5 U	2 J	5 U	5 U	5 U	5 U	5 U	30	5 U	5 U
MW-10	9.2	5 U	5 U	5 U	1 J	5 U	5 U	5 U	5 U	5 U	18	5 U	5 U
MW-10	11.2	5 U	5 U	5 U	1 J	1 J	5 U	5 U	5 U	5 U	17	5 U	5 U
MW-10	13.2	5 U	5 U	5 U	1 J	5 U	5 U	5 U	5 U	5 U	13	5 U	5 U
MW-10	15.2	5 U	5 U	5 U	0 J	5 U	5 U	5 U	5 U	5 U	10	5 U	5 U
MW-10	17.2	5 U	5 U	5 U	0 J	5 U	5 U	5 U	5 U	5 U	9	5 U	5 U
MW-10	18.9	5 U	5 U	5 U	0 J	5 U	5 U	5 U	5 U	5 U	6	5 U	5 U
MW-12	30.5	94 J	73	1,500	970 D	99 J	77	100 U	4 J	1,700	1,400 D	100 U	5 U
MW-12	32.2	91 J	52	1,500	510 D	96 J	57	100 U	2 J	1,700	800 D	100 U	5 U
MW-12	33.7	91 J	87	1,500	780 D	93 J	140	100 U	2 J	1,900	960 D	100 U	5 U
MW-12*	35.1	86 J	260	1,500	2,200	100	450	100 U	100 U	1,800	2,100	100 U	100 U
MW-12	37.0	230	890	2,600	3,700	500	1,500	100 U	500 U	2,400	3,800	100 U	500 U
MW-12	38.5	1,600	1,800	8,800	7,900	2,600	3,000	500 U	500 U	7,800	7,600	500 U	500 U

Table 5. Concentrations of selected chlorinated volatile organic compounds in water from diffusion and low-flow sampling, Naval Air Station North Island, California, January 2000—Continued

[µg/L, micrograms per liter; D, sample was diluted; U, value was below the analytical quantitation limit; J, estimated value; NA, not applicable; E, the detected result is between the sample-specific estimated quantitation and the method detection limit; *, sample collected by using bladder pump; #, data from OHM Remediation Services Corporation (2000)]

Well	Depth (feet)	1,1-Dichloroethane (µg/L)		1,1-dichloroethene (µg/L)		Cis-1,2-dichloroethene (µg/L)		Tetrachloroethene (µg/L)		Trichloroethene (µg/L)		Vinyl chloride (µg/L)		
		Diffusion	Low flow	Diffusion	Low flow	Diffusion	Low flow	Diffusion	Low flow	Diffusion	Low flow	Diffusion	Low flow	Diffusion
MW-13A	6.5	5 U	5 U	5 U	5 U	39	47	5 U	5 U	5 U	5 U	5 U	4 J	6
MW-13A	8.0	5 U	5 U	5 U	5 U	78	46	5 U	5 U	5 U	5 U	5 U	9	6
MW-13A	9.4	5 U	5 U	5 U	5 U	77	47	5 U	5 U	5 U	5 U	5 U	9	6
MW-13A	10.9	5 U	5 U	5 U	5 U	74	53	5 U	5 U	5 U	5 U	5 U	8	7
MW-13A	12.4	5 U	5 U	5 U	5 U	46	55	5 U	5 U	5 U	5 U	5 U	5	7
MW-13A*#	12.0	NA	5 U	NA	5 U	61	61	NA	5 U	NA	5 U	5 U	NA	7
MW-13B	24.9	5 U	5 U	4 J	4 J	3,100 D	2,600 D	5 U	5 U	5 U	4 J	5	1,900 D	1,900 D
MW-13B	26.2	5 U	5 U	4 J	5 U	2,600 D	2,600 D	5 U	5 U	5 U	4 J	6	2,000 D	1,600 D
MW-13B	27.6	5 U	5 U	5 J	5 J	2,700 D	2,900 D	5 U	5 U	5 U	8	7	2,400 D	1,700 D
MW-13B*#	26.0	NA	5 U	NA	5 U	NA	3,100	NA	5 U	NA	5 U	5 U	NA	1,400
MW-13C	45.5	5 U	5 U	5 U	5 U	2 J	2 J	5 U	5 U	5 U	5 U	5 U	5 U	5 U
MW-13C	46.7	5 U	5 U	5 U	5 U	5 U	3 J	5 U	5 U	5 U	5 U	5 U	5 U	5 U
MW-13C	48.1	5 U	5 U	5 U	5 U	5 U	2 J	5 U	5 U	5 U	5 U	5 U	5 U	5 U
MW-13C*#	46.0	NA	5 U	NA	5 U	NA	1 J	NA	5 U	NA	5 U	5 U	NA	5 U
MW-68A	21.7	7 J	4 J	25 U	5 U	140	81	7 J	2 J	570	320 D	25 U	5 U	5 U
MW-68A	23.0	50 U	5 J	50 U	5 U	160	87	50 U	1 J	730	190	50 U	5 U	5 U
MW-68B	34.5	85 J	5,000 U	5,800	4,700 J	87 J	5,000 U	93 J	5,000 U	49,000 D	28,000	250 U	5,000 U	5,000 U
MW-68B	37.0	5,000 U	5,000 U	7,700	4,400 J	5,000 U	5,000 U	5,000 U	5,000 U	62,000	34,000	5,000 U	5,000 U	5,000 U
MW-68B	38.5	5,000 U	5,000 U	8,300	5,000 J	5,000 U	5,000 U	5,000 U	5,000 U	130,000	62,000	5,000 U	5,000 U	5,000 U
MW-68C	56.0	50 U	50 U	20 J	50 U	50 U	50 U	50 U	50 U	1,400	680	50 U	50 U	50 U
MW-68C	57.5	100 U	50 U	100 U	10 J	100 U	50 U	100 U	50 U	2,700	880	100 U	50 U	50 U
MW-68C	59.0	250 U	50 U	250 U	50 U	250 U	50 U	250 U	50 U	4,100	1,100	250 U	50 U	50 U

Table 5. Concentrations of selected chlorinated volatile organic compounds in water from diffusion and low-flow sampling, Naval Air Station North Island, California, January 2000—Continued

[µg/L, micrograms per liter; D, sample was diluted; U, value was below the analytical quantitation limit; J, estimated value; NA, not applicable; E, the detected result is between the sample-specific estimated quantitation and the method detection limit; *, sample collected by using bladder pump; #, data from OHM Remediation Services Corporation (2000)]

Well	Depth (feet)	1,1-Dichloroethane (µg/L)		1,1-Dichloroethene (µg/L)		<i>cis</i> -1,2-Dichloroethene (µg/L)		Tetrachloroethene (µg/L)		Trichloroethene (µg/L)		Vinyl chloride (µg/L)	
		Diffusion	Low flow	Diffusion	Low flow	Diffusion	Low flow	Diffusion	Low flow	Diffusion	Low flow	Diffusion	Low flow
MW-68C2	37.3	1,000 U	500 U	2,400	1,200	490 J	360 J	1,000 U	500 U	19,000	9,100	1,000 U	500 U
MW-68C2	39.1	2,500 U	2,500 U	4,100	3,400	1,000 J	2,500 U	2,500 U	2,500 U	47,000	34,000	2,500 U	2,500 U
MW-68C2	40.5	5,000 U	2,500 U	5,400	2,700	5,000 U	2,500 U	5,000 U	2,500 U	84,000	39,000	5,000 U	2,500 U
MW-68C2	42.1	10,000 U	2,500 U	14,000	2,800	10,000 U	2,500 U	10,000 U	2,500 U	200,000	46,000	10,000 U	2,500 U
MW-68C2	44.2	10,000 U	2,500 U	7,800 J	3,600	10,000 U	2,500 U	10,000 U	2,500 U	110,000	54,000	10,000 U	2,500 U
MW-68C2	46.1	10,000 U	1,000 U	7,500 J	1,200	10,000 U	1,000 U	10,000 U	1,000 U	110,000	17,000	10,000 U	1,000 U
MW-68C2	47.9	10,000 U	5,000 U	7,400 J	4,300 J	10,000 U	5,000 U	10,000 U	5,000 U	110,000	55,000	10,000 U	5,000 U
MW-68C2	49.9	10,000 U	2,500 U	7,200 J	4,400	10,000 U	2,500 U	10,000 U	2,500 U	100,000	53,000	10,000 U	2,500 U
MW-68C2	52.0	2,500 U	2,500 U	3,000	2,800	2,500 U	2,500 U	2,500 U	2,500 U	42,000	42,000	2,500 U	2,500 U
MW-68C2	53.9	1,000 U	2,500 U	1,100	2,500 J	1,000 U	2,500 U	1,000 U	2,500 U	14,000	38,000	1,000 U	2,500 U
MW-68C2	55.6	500 U	1,000 U	600	1,000	500 U	1,000 U	500 U	1,000 U	8,800	14,000	500 U	1,000 U
MW-68C2	57.5	500 U	500 U	450 J	710	500 U	500 U	500 U	500 U	7,300	9,100	500 U	500 U
MW-68C2	59.5	500 U	500 U	350 J	920	500 U	500 U	500 U	500 U	7,000	11,000	500 U	500 U
MW-68C2	61.5	500 U	500 U	280 J	740	500 U	500 U	500 U	500 U	6,500	11,000	500 U	500 U
PW-15	25.4	45	53	4 J	12	42	200 E	5 U	2 J	4 J	15	82	66
PW-15	27.1	52	52	8	22	130	850 D	1 J	8	7	92	72	29
PW-15	28.5	73 J	77 J	36 J	66 J	1,500	2,200	100 J	250 U	180	2,500	100 U	250 U
PW-15	30.2	97 J	500 U	250 U	500 U	2,900	1,600	59 J	500 U	3,000	4,700	250 U	500 U
PW-15	31.8	500 U	500 U	500 U	500 U	1,900	1,800	500 U	500 U	5,500	5,800	500 U	500 U
PW-15	33.2	500 U	500 U	500 U	500 U	2,400	2,100	120 J	130 J	7,500	7,300	500 U	500 U
PW-55	27.1	2,500 U	2,500 U	2,500 U	2,500 U	7,300	5,700	2,500 U	2,500 U	39,000	31,000	2,500 U	2,500 U
PW-55	28.9	2,500 U	2,500 U	2,500 U	2,500 U	6,500	5,900	2,500 U	2,500 U	39,000	32,000	2,500 U	2,500 U
PW-55	30.6	2,500 U	2,500 U	2,500 U	2,500 U	6,600	5,900	2,500 U	2,500 U	38,000	34,000	2,500 U	2,500 U

Table 5. Concentrations of selected chlorinated volatile organic compounds in water from diffusion and low-flow sampling, Naval Air Station North Island, California, January 2000—Continued

[µg/L, micrograms per liter; D, sample was diluted; U, value was below the analytical quantitation limit; J, estimated value; NA, not applicable; E, the detected result is between the sample-specific estimated quantitation and the method detection limit; *, sample collected by using bladder pump; #, data from OHM Remediation Services Corporation (2000)]

Well	Depth (feet)	1,1-Dichloroethane (µg/L)		1,1-Dichloroethene (µg/L)		<i>cis</i> -1,2-Dichloroethene (µg/L)		Tetrachloroethene (µg/L)		Trichloroethene (µg/L)		Vinyl chloride (µg/L)	
		Diffusion	Low flow	Diffusion	Low flow	Diffusion	Low flow	Diffusion	Low flow	Diffusion	Low flow	Diffusion	Low flow
PW-55	31.9	2,500 U	2,500 U	2,500 U	2,500 U	6,800	5,300	2,500 U	2,500 U	38,000	29,000	2,500 U	2,500 U
PW-55	33.1	2,500 U	2,500 U	2,500 U	2,500 U	6,300	5,500	2,500 U	2,500 U	33,000	29,000	2,500 U	2,500 U
PW-66	25.5	1,000 U	500 U	1,000 U	38 J	2,000	1,600	1,000 U	599 U	17,000	13,000	1,000 U	500 U
PW-66	27.3	500 U	500 U	500 U	500 U	3,400	1,000	500 U	500 U	9,000	6,400	500 U	500 U
PW-66	29.1	50 U	500 U	38 J	500 U	23 J	500 U	50 U	500 U	770	9,800	50 U	500 U
PW-66	30.8	25 U	500 U	49	500 U	25 U	500 U	25 U	500 U	180	5,600	25 U	500 U
PW-66	32.3	5 U	500 U	72	500 U	5 U	500 U	5 U	500 U	48	6,200	5 U	500 U
PW-66	33.9	5 U	500 U	130	500 U	5 U	500 U	1 J	500 U	13	6,000	5 U	500 U
S2-MW-6A	6.5	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U
S2-MW-6A	7.9	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	2 U	5 U	5 U
S2-MW-6A	9.2	5 U	5 U	5 U	5 U	5 U	5						
S2-MW-6A	10.6	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	2 J	2 J	5 U	5 U
S2-MW-6A	12.0	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	2 J	5 U	5 U
S2-MW-6A	13.3	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	1 J	5 U	5 U	5 U
S2-MW-6A	14.7	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	2 J	5 U	5 U
S2-MW-6A	16.1	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	2 J	5 U	5 U
S2-MW-6A	17.5	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	2 J	5 U	5 U
S2-MW-6A	19.0	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U

Table 6. Concentrations of benzene, ethylbenzene, toluene, and total xylenes in water from diffusion and low-flow sampling, Naval Air Station North Island, California, January 2000

[µg/L, micrograms per liter; U, value was below the analytical quantitation limit; J, estimated value; NA, not applicable; *, sample collected by using bladder pump; #, data from OHM Remediation Services Corporation (2000)]

Well	Depth (feet)	Benzene (µg/L)		Ethylbenzene (µg/L)		Toluene (µg/L)		Total xylenes (µg/L)	
		Diffusion	Low flow	Diffusion	Low flow	Diffusion	Low flow	Diffusion	Low flow
MW-5D	50.8	50 U	25 U	50 U	25 U	50 U	25 U	50 U	25 U
MW-5D	52.3	250 U	50 U	250 U	50 U	250 U	50 U	250 U	50 U
MW-5D	54.2	50 U	100 U	50 U	100 U	50 U	100 U	50 U	100 U
MW-5D*	55.8	5 U	25 U	5 U	25 U	5 U	25 U	5 U	25 U
MW-5D	57.4	50 U	5 U	50 U	5 U	50 U	5 U	50 U	5 U
MW-5D	59.0	50 U	5 U	50 U	5 U	50 U	5 U	50 U	5 U
MW-9*	27.6	250 U	250 U	250 U	250 U	250 U	250 U	250 U	250 U
MW-10	7.8	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U
MW-10	9.2	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U
MW-10	11.2	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U
MW-10	13.2	5 U	5 U	5 U	5 U	5 U	5 U	1 J	5 U
MW-10	15.2	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U
MW-10	17.2	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U
MW-10	18.9	5 U	5 U	5 U	5 U	5 U	5 U	5 U	25 J
MW-12	30.5	100 U	5 U	100 U	5 U	100 U	5 U	100 U	5 U
MW-12	32.2	100 U	5 U	100 U	5 U	100 U	5 U	100 U	5 U
MW-12	33.7	100 U	5 U	100 U	5 U	100 U	5 U	100 U	5 U
MW-12*	35.1	100 U	100 U	100 U	100 U	100 U	100 U	100 U	100 U
MW-12	37.0	100 U	500 U	100 U	120 U	100 U	500 U	100 U	360 U
MW-12	38.5	500 U	500 U	500 U	500 U	500 U	500 U	500 U	500 U
MW-13A	6.5	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U
MW-13A	8.0	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U
MW-13A	9.4	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U
MW-13A	10.9	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U
MW-13A	12.4	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U
MW-13A*##	12.0	NA	5 U	NA	5 U	NA	5 U	NA	
MW-13B	24.9	9	5	5 U	5 U	5 U	5 U	5 U	5 U
MW-13B	26.2	5	4 J	5 U	5 U	5 U	5 U	5 U	5 U
MW-13B	27.6	1 J	4 J	5 U	5 U	1 J	5 U	5 U	5 U
MW13B*##	26.0	NA	5 U	NA	5 U	NA	5 U	NA	5 U
MW-13C	45.5	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U
MW-13C	46.7	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U

Table 6. Concentrations of benzene, ethylbenzene, toluene, and total xylenes in water from diffusion and low-flow sampling, Naval Air Station North Island, California, January 2000—Continued

[µg/L, micrograms per liter; U, value was below the analytical quantitation limit; J, estimated value; NA, not applicable; *, sample collected by using bladder pump; #, data from OHM Remediation Services Corporation (2000)]

Well	Depth (feet)	Benzene (µg/L)		Ethylbenzene (µg/L)		Toluene (µg/L)		Total xylenes (µg/L)	
		Diffusion	Low flow	Diffusion	Low flow	Diffusion	Low flow	Diffusion	Low flow
MW-13C	48.1	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U
MW-13C*#	46.0	NA	5 U	NA	5 U	NA	5 U	NA	5 U
MW-68A	21.7	25 U	5 U	25 U	5 U	25 U	5 U	25 U	5 U
MW-68A	23.0	50 U	5 U	50 U	5 U	50 U	5 U	50 U	5 U
MW-68B	34.5	250 U	5,000 U	250 U	5,000 U	250 U	5,000 U	250 U	5,000 U
MW-68B	37.0	5,000 U	5,000 U	5,000 U	5,000 U	5,000 U	5,000 U	5,000 U	5,000 U
MW-68B	38.5	5,000 U	5,000 U	5,000 U	5,000 U	5,000 U	5,000 U	5,000 U	5,000 U
MW-68C	56.0	50 U	50 U	50 U	50 U	50 U	50 U	50 U	50 U
MW-68C	57.5	100 U	50 U	100 U	50 U	100 U	50 U	100 U	50 U
MW-68C	59.0	250 U	12 J	250 U	50 U	250 U	50 U	250 U	50 U
MW-68C2	37.3	1,000 U	500 U	1,000 U	500 U	1,000 U	500 U	1,000 U	500 U
MW-68C2	39.1	2,500 U	2,500 U	2,500 U	2,500 U	2,500 U	2,500 U	2,500 U	2,500 U
MW-68C2	40.5	5,000 U	2,500 U	5,000 U	2,500 U	5,000 U	2,500 U	5,000 U	2,500 U
MW-68C2	42.1	10,000 U	2,500 U	10,000 U	2,500 U	10,000 U	2,500 U	10,000 U	2,500 U
MW-68C2	44.2	10,000 U	2,500 U	10,000 U	2,500 U	10,000 U	2,500 U	10,000 U	2,500 U
MW-68C2	46.1	10,000 U	1,000 U	10,000 U	1,000 U	10,000 U	1,000 U	10,000 U	1,000 U
MW-68C2	47.9	10,000 U	5,000 U	10,000 U	5,000 U	10,000 U	5,000 U	10,000 U	5,000 U
MW-68C2	49.9	10,000 U	2,500 U	10,000 U	2,500 U	10,000 U	2,500 U	10,000 U	2,500 U
MW-68C2	52.0	2,500 U	2,500 U	2,500 U	2,500 U	2,500 U	2,500 U	2,500 U	2,500 U
MW-68C2	53.9	1,000 U	2,500 U	1,000 U	2,500 U	1,000 U	2,500 U	1,000 U	2,500 U
MW-68C2	55.6	500 U	1,000 U	500 U	1,000 U	500 U	1,000 U	500 U	1,000 U
MW-68C2	57.5	500 U	500 U	500 U	500 U	500 U	500 U	500 U	500 U
MW-68C2	59.5	500 U	500 U	500 U	500 U	500 U	500 U	500 U	500 U
MW-68C2	61.5	500 U	500 U	500 U	500 U	500 U	500 U	500 U	500 U
PW-15	25.4	4 J	2 J	16	9	7	3 J	75	28
PW-15	27.1	3 J	3 J	15	3 J	5 J	5 U	52	7
PW-15	28.5	100 U	250 U	100 U	250 U	100 U	250 U	100 U	250 U
PW-15	30.2	250 U	500 U	250 U	500 U	250 U	500 U	250 U	500 U
PW-15	31.8	500 U	500 U	500 U	500 U	500 U	500 U	500 U	500 U
PW-15	33.2	500 U	500 U	500 U	500 U	500 U	500 U	500 U	500 U
PW-55	27.1	2,500 U	2,500 U	2,500 U	2,500 U	2,500 U	2,500 U	2,500 U	2,500 U
PW-55	28.9	2,500 U	2,500 U	2,500 U	2,500 U	2,500 U	2,500 U	2,500 U	2,500 U
PW-55	30.6	2,500 U	2,500 U	2,500 U	2,500 U	2,500 U	2,500 U	2,500 U	2,500 U

Table 6. Concentrations of benzene, ethylbenzene, toluene, and total xylenes in water from diffusion and low-flow sampling, Naval Air Station North Island, California, January 2000—Continued

[µg/L, micrograms per liter; U, value was below the analytical quantitation limit; J, estimated value; NA, not applicable; *, sample collected by using bladder pump; #, data from OHM Remediation Services Corporation (2000)]

Well	Depth (feet)	Benzene (µg/L)		Ethylbenzene (µg/L)		Toluene (µg/L)		Total xylenes (µg/L)	
		Diffusion	Low flow	Diffusion	Low flow	Diffusion	Low flow	Diffusion	Low flow
PW-55	31.9	2,500 U	2,500 U	2,500 U	2,500 U	2,500 U	2,500 U	2,500 U	2,500 U
PW-55	33.1	2,500 U	2,500 U	2,500 U	2,500 U	2,500 U	2,500 U	2,500 U	2,500 U
PW-66	25.5	1,000 U	500 U	1,000 U	500 U	1,000 U	500 U	1,000 U	500 U
PW-66	27.3	500 U	500 U	500 U	500 U	500 U	500 U	500 U	500 U
PW-66	29.1	50 U	500 U	50 U	500 U	50 U	500 U	50 U	500 U
PW-66	30.8	25 U	500 U	25 U	500 U	25 U	500 U	25 U	500 U
PW-66	32.3	5 U	500 U	5 U	500 U	5 U	500 U	5 U	500 U
PW-66	33.9	5 U	500 U	5 U	500 U	5 U	500 U	5 U	500 U
S2-MW-6A	6.5	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U
S2-MW-6A	7.9	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U
S2-MW-6A	9.2	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U
S2-MW-6A	10.6	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U
S2-MW-6A	12.0	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U
S2-MW-6A	13.3	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U
S2-MW-6A	14.7	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U
S2-MW-6A	16.1	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U
S2-MW-6A	17.5	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U
S2-MW-6A	19.0	5 U	5 U	5 U	5 U	5 U	5 U	5 U	5 U

20 percent, the concentrations obtained using the diffusion samplers were slightly higher than those concentrations obtained using the bladder pump. The concentrations obtained using the diffusion samplers in well MW-13B were slightly higher, but similar to the concentrations obtained using the peristaltic pump (fig. 3). Concentrations of toluene and total xylenes were present in water obtained from both the diffusion samplers and the peristaltic pump (fig. 3); toluene and total xylenes were not detectable (less than 5 µg/L) in water from the bladder pump. The data suggest that the diffusion samplers performed equally well with the bladder pump in wells MW-13A and MW-13B for cDCE. The higher concentrations of vinyl chloride, toluene, and total xylenes in water from the diffusion samplers relative to water from the bladder pump indicate that the diffusion samplers obtained more discrete samples from these wells; however, disturbing the well water by using the peristaltic pump and removing the

diffusion samplers prior to sampling with the bladder pump may have induced mixing and affected the quality of the water sampled by the bladder pump.

Comparison of Diffusion-Sampler Results to Peristaltic-Pump Results

The remaining comparisons between diffusion-sampler and low-flow sampler methods utilized multiple diffusion-sampling and low-flow sampling points within screened intervals. At most depths, low-flow sampling was conducted by using peristaltic pumps. In contrast to bladder pumps, using peristaltic pumps in some wells potentially could cause degassing of samples during recovery, which could result in underestimating actual VOC concentrations. Thus, VOC concentrations in water obtained using peristaltic

Table 7. Comparison of concentrations of selected volatile organic compounds in water from a diffusion sampler and in water from low-flow purging using a bladder pump at the same depth, Naval Air Station North Island, California, January 2000

[*, average percent difference; -, concentration measured in diffusion sampler was lower than concentration measured in low-flow sample]

Constituent	Diffusion samples		Low-flow bladder-pump samples		Percent difference
	Depth, in feet below land surface	Concentration, in micrograms per liter	Depth, in feet below land surface	Concentration, in micrograms per liter	
Well MW-9					
1,1-Dichloroethene (1,1-DCE)	31	4,000	31	3,500	2.0
Tetrachloroethene (PCE)	31	260	31	330	-21.0
Trichloroethene (TCE)	31	3,300	31	3,200	3.0
Well MW-5					
Trichloroethene (TCE)	55.75	360	55.75	280, 320	17*
Well MW-12					
<i>cis</i> -1,2-Dichloroethene (cDCE)	35.1	100	35.1	450	-78
Trichloroethene (TCE)	35.1	1,800	35.1	2,100	-14
Well MW-13A					
<i>cis</i> -1,2-Dichloroethene (cDCE)	10.9 - 12.4	46 - 74	12	61	Within range
Vinyl chloride	10.9 - 12.4	5 - 8	12	7.4	Within range
Well MW-13B					
<i>cis</i> -1,2-Dichloroethene (cDCE)	24.85 - 26.15	3,100 - 2,600	26	3,100	Within range
Toluene	24.85 - 26.15	9	26	<5	Not applicable
Total xylenes	24.85 - 26.15	111 - 110	26	<5	Not applicable
Vinyl chloride	24.85 - 26.15	1,900 - 2,000	26	1,600	18*

pumps may be representative of concentrations in ground water at some wells but may underestimate actual concentrations in ground water at other wells. Moreover, when multiple depths within a screened interval are purged using low-flow methods, there is a potential for each low-flow sampling event to disturb the equilibrated water column. If the pumping rate during low-flow sampling is low enough to prevent drawdown in the well, then all of the pumped water is replaced by ground water from the aquifer; however, the zone of influence contributing water to the well may not be adjacent to the pump. Thus, in a chemically stratified screened interval where multiple depth intervals are sequentially sampled, water entering the well screen from early low-flow samplings may influence concentrations obtained in later samplings as a result of vertical transport and mixing in the well screen. Despite these uncertainties, the use of multiple-level low-flow sampling methods using

peristaltic pumps sometimes can provide an estimate of contaminant vertical distribution in the screened interval, which can be used as a comparison for the diffusion samplers.

In most of the observation wells, the vertical concentration gradients obtained using the diffusion-sampler and low-flow sampler methods were similar. However, in several cases, the concentrations in water obtained by using the peristaltic pump were lower than the concentrations in water obtained by using the diffusion samplers (figs. 4, 5, and 6). An example of this is TCE concentrations measured in water from wells MW-68A, MW-68B, and MW-68C. TCE concentrations were approximately 43 to 73 percent lower in water samples collected by using low-flow sampling methods and peristaltic pumps than in samples collected by the diffusion samplers. This substantial difference in concentrations between the two methods is expected if VOCs were lost by degassing as a result of

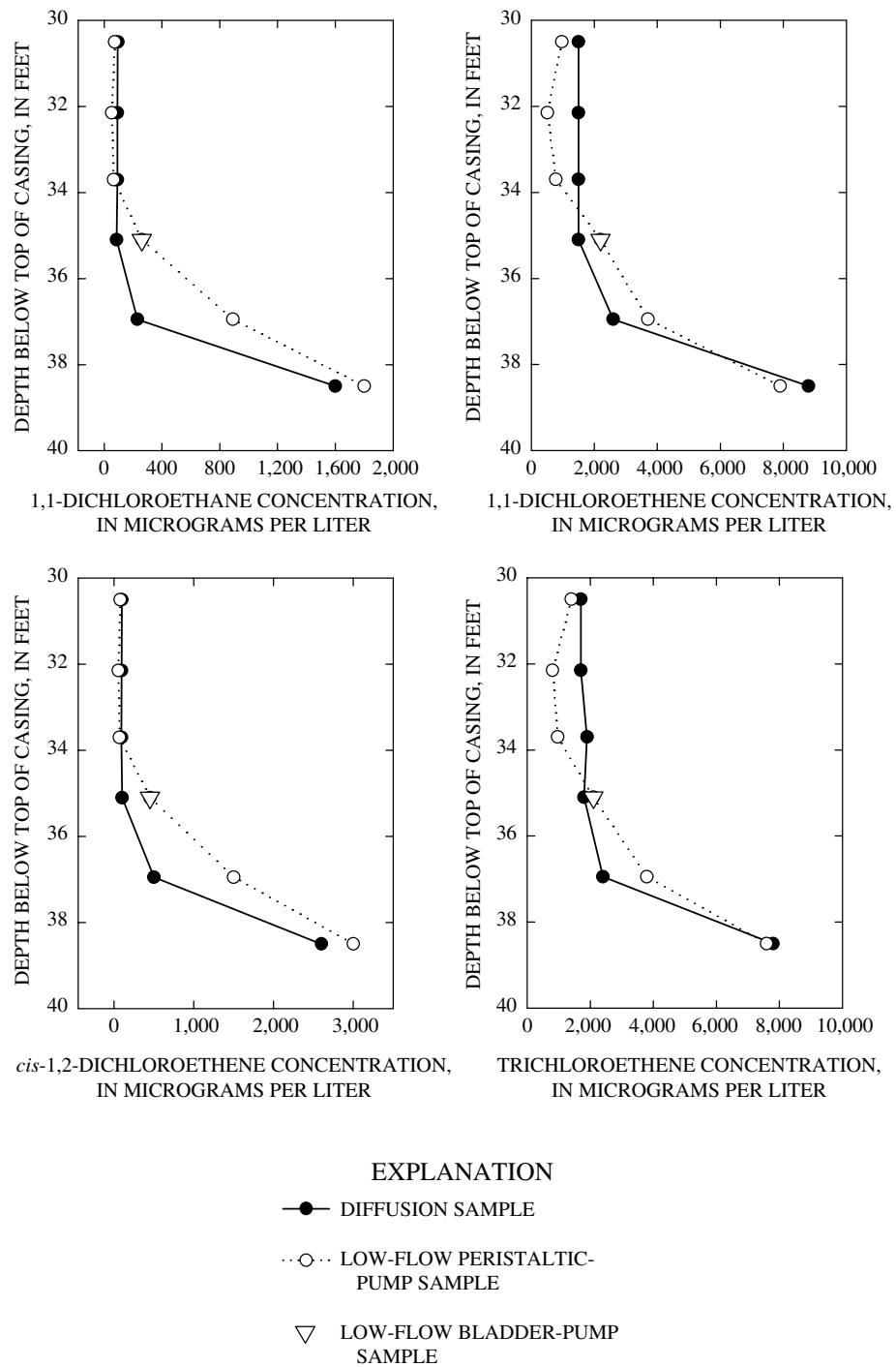


Figure 2. Comparison of diffusion and low-flow samples in ground water at well MW-12, Naval Air Station North Island, California, January 2000.

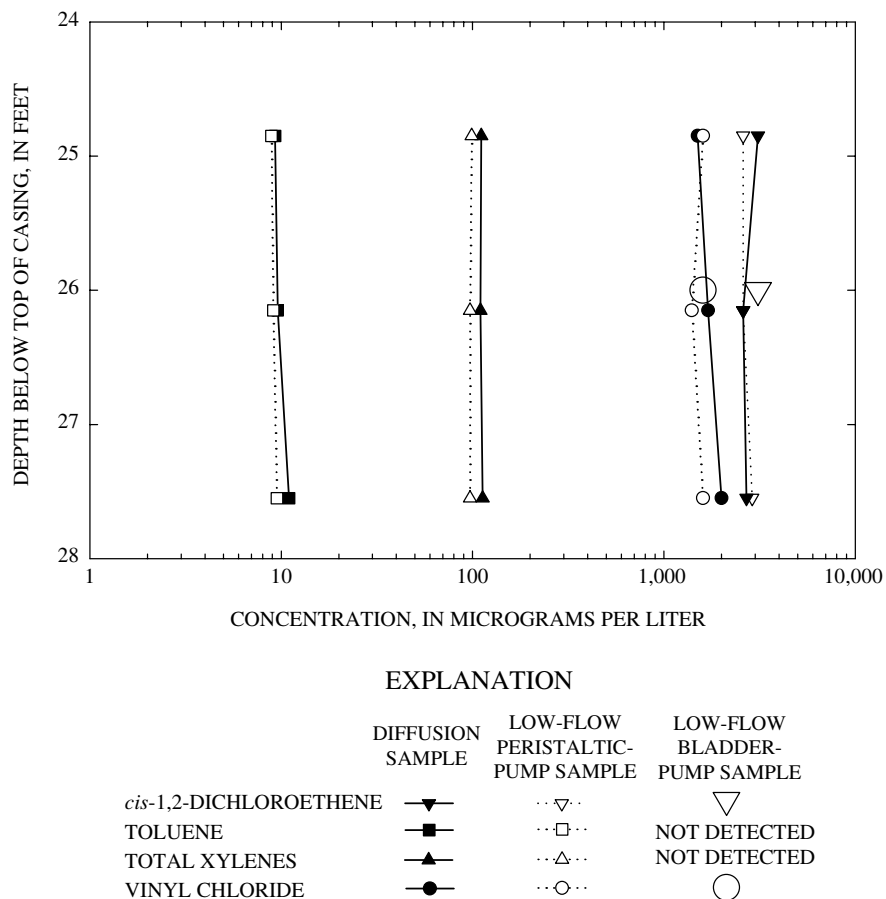


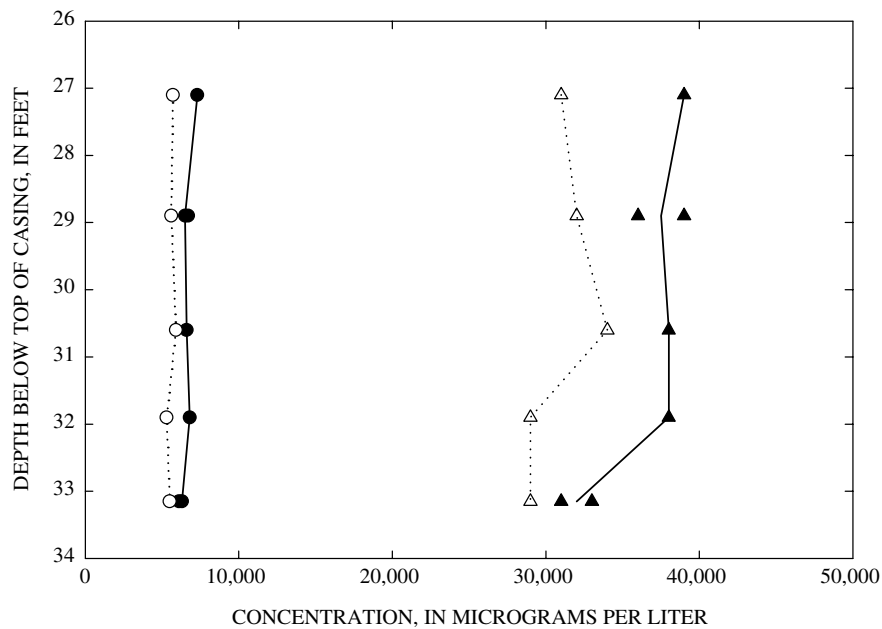
Figure 3. Comparison of diffusion and low-flow samples in ground water at well MW-13B, Naval Air Station North Island, California, January 2000.

using peristaltic pumps or if mixing in the well screens occurred during pumping. The vertical concentration distribution between the two methods implies that the VOC concentrations measured in water from diffusion samplers reflected the vertical distribution of contaminants in the aquifer adjacent to the screened interval more accurately than the peristaltic-pump sampling.

Further comparison of TCE concentration data from the two sampling methods indicates that diffusion sampling provides a point sample, whereas sequential low-flow sampling of multiple horizons within a single well screen can induce mixing. In general, the vertical sequence of low-flow sampling in the wells began with the shallowest depth interval and ended with the deepest interval. In well PW-66, TCE data show that concentrations in water collected with a diffusion sampler were highest in the shallowest sampled depth, and then decreased sharply over the 5-ft depth interval below this shallowest depth (fig. 7).

Although the highest TCE concentration obtained by low-flow sampling also was at the shallowest horizon, it was approximately 24 percent lower than the concentration obtained from the corresponding diffusion sampler, and the vertical stratification was less sharply defined. These data suggest that as low-flow sampling with a peristaltic pump progressed vertically downward, the pumping gradually mixed the TCE-contaminated water from the shallowest sampling depth with water from deeper intervals, thus obscuring the original contaminant stratification (fig. 7).

A similar effect can be seen in the data from wells MW-12 and PW-15 (figs. 2 and 8). At these wells, the shallowest interval was relatively uncontaminated. The comparison between diffusion samples and low-flow samples at this shallowest depth showed a relatively close match between *c*DCE and TCE concentrations. However, as sampling progressed vertically downward toward the interface of the



EXPLANATION

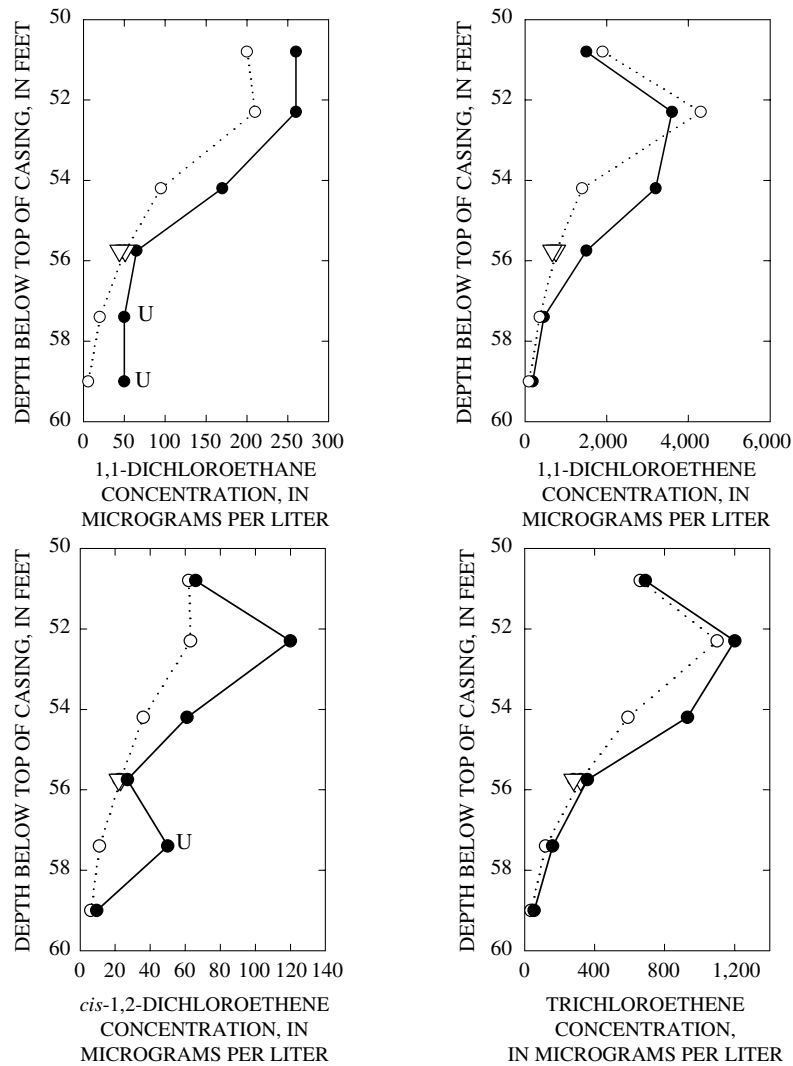
	DIFFUSION SAMPLE	LOW-FLOW SAMPLE
<i>cis</i> -1,2-DICHLOROETHENE	●—	··○··
TRICHLOROETHENE	▲—	··△··

Figure 4. Comparison of diffusion and low-flow samples in ground water at well PW-55, Naval Air Station North Island, California, January 2000.

stratified contamination, the low-flow sample concentrations generally increased higher than the diffusion-sample concentrations, which is to be expected if the zone of influence for the low-flow pumping captured the more contaminated ground water in the well. In general, the data suggest that diffusion sampling provides a more precise delineation of the contaminant stratification within the screened interval than low-flow sampling.

Insight into the use of diffusion samplers in a chemically stratified screened interval can be observed in the data from wells at the MW-68 cluster (figs. 6D and 6E). Unlike the other wells, two peristaltic pumps were used to low-flow sample well MW-68C2. Starting simultaneously from both the uppermost and the lowermost sample depths, sampling progressed sequentially toward the center of the 25-ft screened interval. Results from both the diffusion samples and the low-flow samples showed that the uppermost and lowermost parts of the screened interval were relatively uncontaminated. Concentration data from the

diffusion samples show that substantially higher TCE concentrations occurred between depths of approximately 40 to 50 ft, with a sharp peak at about 42 ft (fig. 6D). Thus, the first water pulled into the well screen from both ends of the screen was relatively uncontaminated. As the low-flow sampling progressed toward the center of the screened interval, the correlation between concentrations obtained from the diffusion samples began to differ substantially from those obtained by low-flow sampling (fig. 6D). Between the depths of approximately 40 to 50 ft, TCE concentrations from low-flow sampling were approximately 47 to 84 percent lower than TCE concentrations from diffusion samplers; additionally, the low-flow sampling data did not indicate a TCE peak concentration at a depth of 42 ft as shown by the diffusion sampling data. A probable explanation for the concentration differences between the two methods is that initially, relatively uncontaminated water was pumped into the screened interval, thus mixing the ground water in the well and diluting concentrations of TCE. As a result,



EXPLANATION

- DIFFUSION SAMPLE
- LOW-FLOW PERISTALTIC-PUMP SAMPLE
- ▽ LOW-FLOW BLADDER-PUMP SAMPLE
- U VALUE WAS BELOW THE ANALYTICAL QUANTITATION LIMIT

Figure 5. Comparison of diffusion and low-flow samples in ground water at well MW-5, Naval Air Station North Island, California, January 2000.

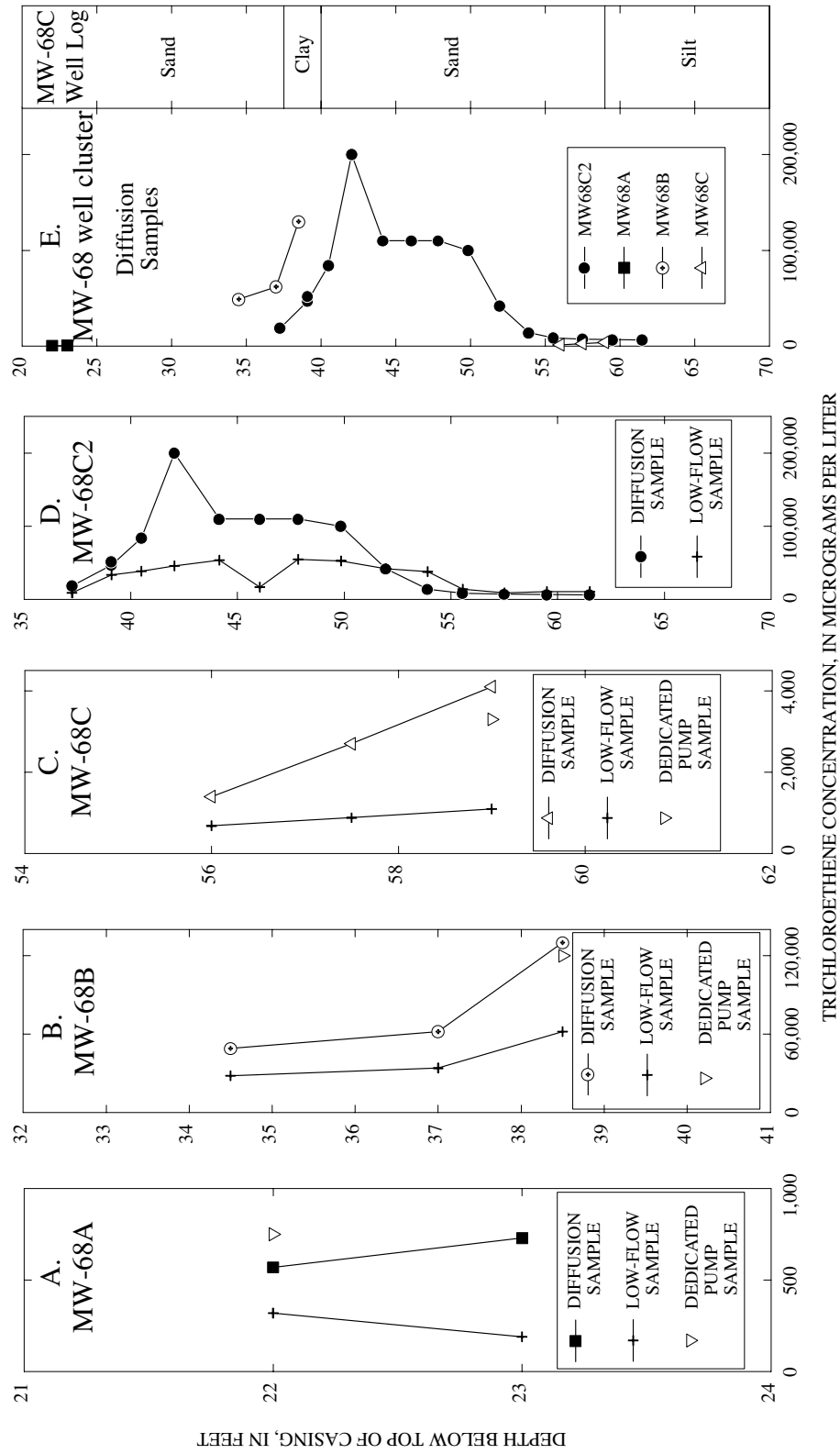


Figure 6. Comparison of diffusion and low-flow samples in ground water at wells (A) MW-68A, (B) MW-68B, (C) MW-68C, and (D) MW-68C2, and (E) comparison of diffusion samples from multiple wells to geologic log of well MW-68C, Naval Air Station North Island, California, January 2000.

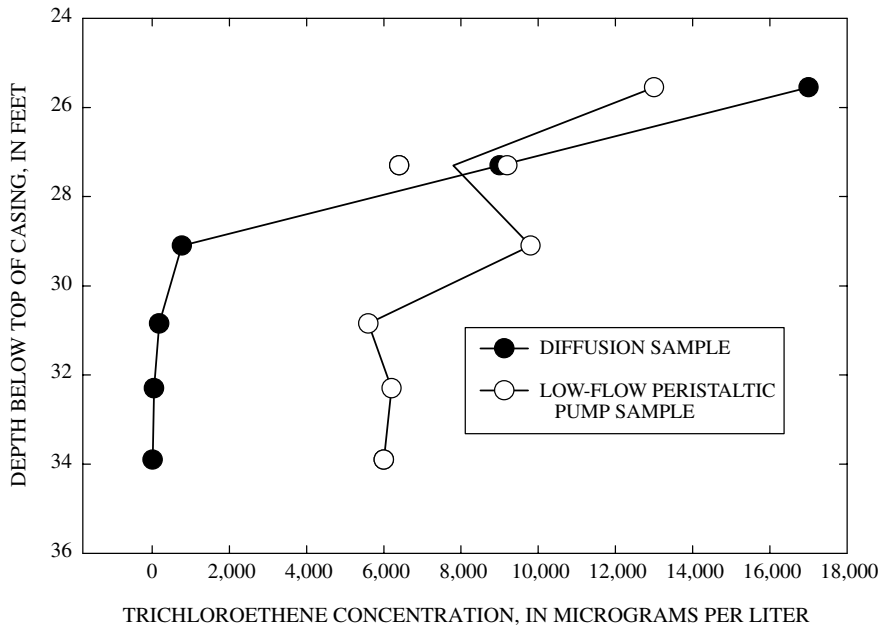


Figure 7. Comparison of trichloroethene concentrations in diffusion and low-flow samples in ground water at well PW-66, Naval Air Station North Island, California 2000.

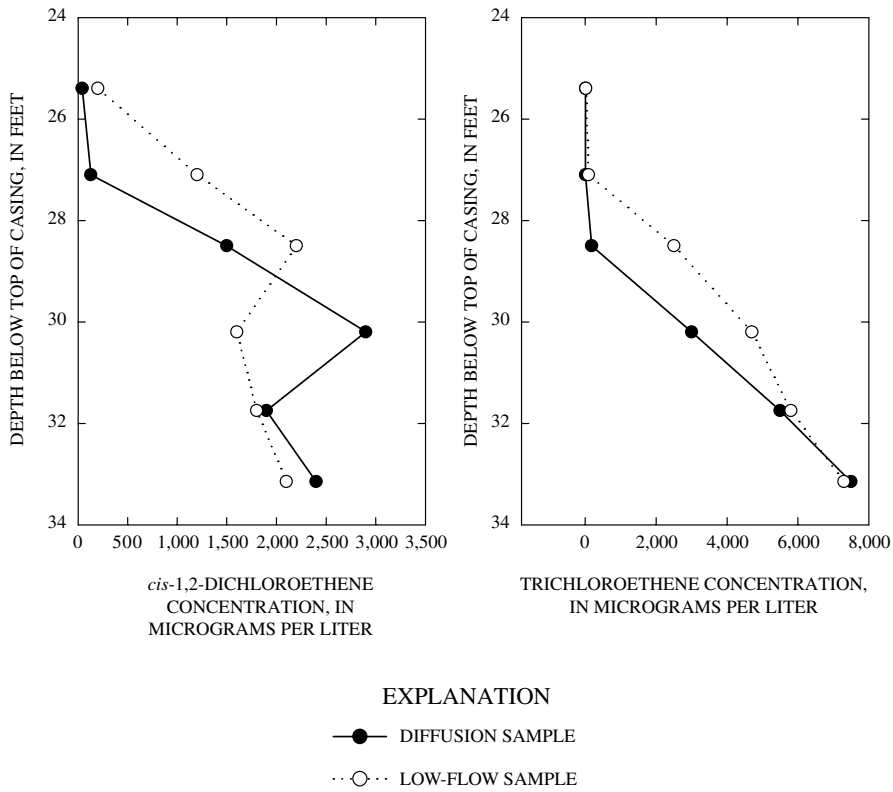


Figure 8. Comparison of diffusion and low-flow samples in ground water at well PW-15, Naval Air Station North Island, California, January 2000.

TCE concentrations were lowered in ground water collected from subsequently sampled depths. Additional VOC losses by degassing during the use of peristaltic pumps probably resulted in further concentration differences between the two sampling methods.

TCE concentration data in diffusion samplers collected from wells MW-68B and MW-68C and concentration data in diffusion samplers collected from adjacent well MW-68C2 support the vertical distribution indicated by the diffusion samplers in well MW-68C2 (figs. 6B, 6C, and 6D). Diffusion samplers from well MW-68C2 indicate that the lowest concentrations in the screened interval are below a depth of approximately 55 ft, and the detected concentrations are similar to those from the same depth in the adjacent well MW-68C (fig. 6E).

Similarly, diffusion samplers from wells MW-68C2 and MW-68B both indicate TCE concentrations increasing with depth between approximately 35 and 40 ft (fig. 6E). The TCE concentrations in diffusion samples from well MW-68B are higher than those from the corresponding depth in well MW-68C2 (fig. 6E). The reasons for the concentration difference between wells MW-68C2 and MW-68B are not known; however, two explanations can be postulated. One explanation is that the contaminant concentrations in well MW-68C2 may have been shifted downward as a result of a vertical hydraulic gradient within the well. Water-level measurements are not shown for well MW-68C2 because they would reflect only compositing across the screened interval; however, evidence for such a hydraulic gradient can be seen in the water-level data for wells MW-68B and MW-68C. The water level in well MW-68B is 0.34 ft higher than the water level in well MW-68C, indicating a net downward hydraulic gradient between the two depths (table 1). Water levels remeasured in March 2000 confirmed the hydraulic gradient. Because well MW-68C2 is only about 5 ft from wells MW-68B and MW-68C, and because the screened interval of well MW-68C2 hydraulically connects the depths sampled by wells MW-68B and MW-68C, the probability is high that there also is a downward hydraulic gradient within well MW-68C2. An alternative explanation is that lithologic heterogeneities in the screened zone place the contamination at slightly different depths in different wells. Evidence for such heterogeneity is the clay layer at a depth of 37.5 to 40 ft in well MW-68C (driller's log, Richard Wong, OHM Remediation, written commun., 2000). Despite the uncertainty regarding

concentration differences between wells, the diffusion samplers appear to have been successful in approximately locating the zone of highest concentrations between the depths of 37 to 52 ft (fig. 6E).

VOC concentrations in water collected from well MW-13A varied less and generally were lower for peristaltic pump sampling compared to diffusion sampling (fig. 9). Following low-flow sampling using a peristaltic pump, well MW-13A was immediately resampled by low-flow sampling using a bladder pump. Although subject to the same mixing potential as the peristaltic pump, the bladder pump has less potential for volatilization loss than the peristaltic pump, and thus, probably provides a more representative sample than the peristaltic pump. The concentrations of *c*DCE and TCE in water obtained using low-flow sampling methods with a bladder pump approximated the average of concentrations obtained in water from the diffusion samplers directly above and below the bladder pump (fig. 8). These findings suggest that data obtained by using the diffusion samplers provided depth-specific VOC concentrations while the data from low-flow sampling represented a mixing of waters in well MW-13A.

In well MW-10, low-flow peristaltic-pump sampling detected low concentrations (30 µg/L or less) of TCE, whereas diffusion sampling detected none (table 5). This difference in concentrations is unusual because the potential for volatilization loss using the peristaltic pump usually results in underestimating ambient concentrations, while diffusion samplers are capable of producing representative samples even at low (less than 20 µg/L) concentrations. According to historical data (OHM Remediation Services Corporation, 2000), TCE has never previously been detected in well MW-10 (sampling dates July 1998, March 1999, June 1999, and September 1999). Furthermore, a resampling of the well using low-flow methodology at multiple horizons in February 2000 also showed that TCE was not present. Thus, it seems that the diffusion samplers accurately reflected VOC concentrations in ground water; the source of low TCE concentrations found in water obtained from low-flow, peristaltic-pump sampling is unknown, but may represent a cross-contamination source not related to local ground water.

Wells S2-MW-06A and MW-13C contained no detectable VOCs (less than 5 µg/L) in water from either the diffusion samples or from the low-flow samples. Thus, the construction materials used in the

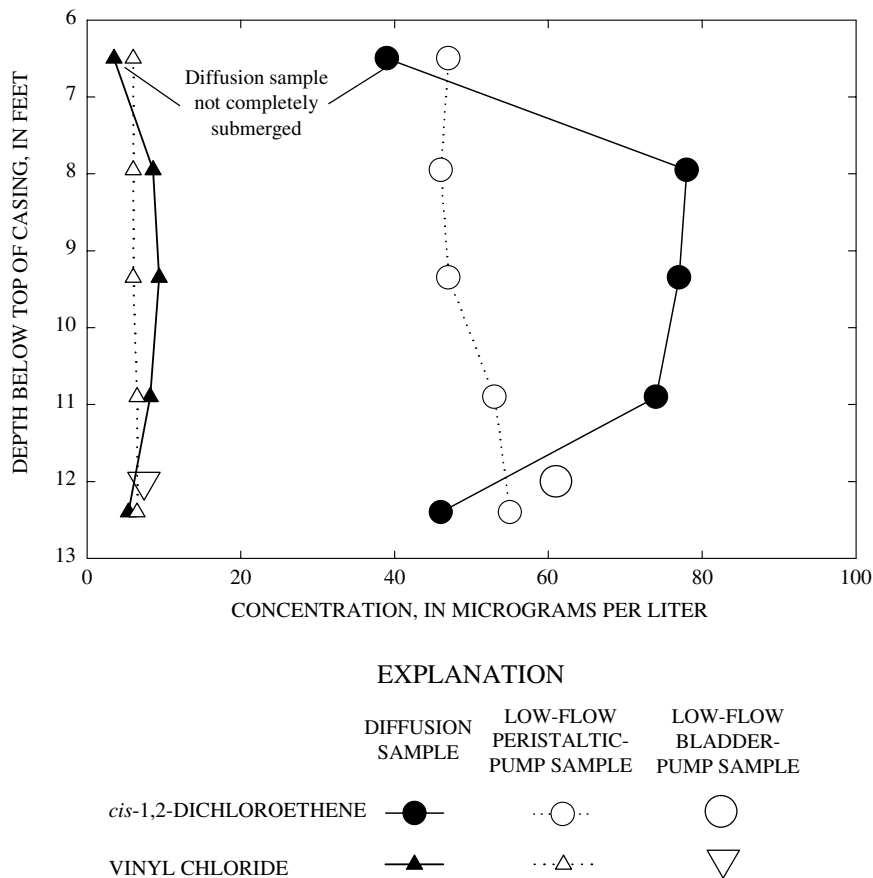


Figure 9. Comparison of diffusion and low-flow samples in ground water well MW-13A, Naval Air Station North Island California, January 2000.

diffusion samplers did not contribute contaminants to the water.

the free-phase fuel in ground water from well MW-11 also contained TCE (table 5).

Diffusion Samplers in Free-Phase Fuel

The diffusion samplers deployed in buckets containing free-phase JP-5 and Stoddard solution from wells MW-11 and PW-17 did not show evidence of structural integrity loss during the 2 months of equilibration. The VOCs detected in the free-phase fuel also were detected in the water from the diffusion samplers (table 8). The VOC concentrations in water from the diffusion samplers were lower than the VOC concentrations in the fuel; however, this is to be expected because the first is an aqueous solution and the second is an organic solvent concentration. The diffusion samplers provided an alternative method for showing that

Contaminant Stratification in Well Screens

The data from this investigation show that substantial stratification of VOCs can be present within a 10-ft well screen. At four observation wells (MW-12, MW-5, PW-66, and PW-15), the data showed a sharp layering of VOCs within the screened interval (figs. 2, 5, 7, and 8). The diffusion-sampler data show that the vertical change in TCE concentrations over a distance of about 5 ft was approximately 17,500 $\mu\text{g/L}$ in well PW-66, approximately 7,300 $\mu\text{g/L}$ in well PW-15, and approximately 5,900 $\mu\text{g/L}$ in well MW-12. At well MW-5, the 1,1-DCE concentration changed by 3,410 $\mu\text{g/L}$, and the TCE concentration changed by

Table 8. Concentrations of selected volatile organic compounds in free-phase jet fuel (JP-5) removed from ground water and in water from diffusion samplers deployed in a bucket containing the free-phase fuel, Naval Air Station North Island, California, January 2000

[$\mu\text{g/L}$, micrograms per liter; J, estimated value; U, value was below the analytical quantitation limit]

Sample source	1,1-Dichloroethane ($\mu\text{g/L}$)		1,1-Dichloroethene ($\mu\text{g/L}$)		<i>cis</i> -1,2-Dichloroethene ($\mu\text{g/L}$)		Tetrachloroethene ($\mu\text{g/L}$)		Trichloroethene ($\mu\text{g/L}$)	
	Diffusion sampler water	Free-phase fuel	Diffusion sampler water	Free-phase fuel	Diffusion sampler water	Free-phase fuel	Diffusion sampler water	Free-phase fuel	Diffusion sampler water	Free-phase fuel
Free product from well PW-17	4 J	5,000 U	5 U	5,000 U	3.9 J	5,000 U	5 U	5,000 U	2 J	5,000 U
Free product from well MW-11	10 U	5,000 U	10 U	5,000 U	10 U	5,000 U	7 J	4,300 J	65	5,200
	Vinyl chloride ($\mu\text{g/L}$)		Benzene ($\mu\text{g/L}$)		Ethylbenzene ($\mu\text{g/L}$)		Toluene ($\mu\text{g/L}$)		Total xylenes ($\mu\text{g/L}$)	
	Diffusion sampler water	Free-phase fuel	Diffusion sampler water	Free-phase fuel	Diffusion sampler water	Free-phase fuel	Diffusion sampler water	Free-phase fuel	Diffusion sampler water	Free-phase fuel
Free product from well PW-17	5 U	5,000 U		5,000 U	70	21,000	112	1,100 J	350	100,000
Free product from well MW-11	10 U	5,000 U	10 U	5,000 U	13	5,700	10 U	5,000 U	120	43,000

1,145 $\mu\text{g/L}$ over a vertical distance of about 7 ft (fig. 5). The concentrations decreased with depth at some wells [MW-5 and PW-66 (figs. 5 and 7, respectively)] and increased with depth at others [MW-12 and PW-15 (figs. 2 and 8, respectively)].

The presence of contaminant stratification in well screens has importance for ground-water sampling. In an environment with a sharp concentration gradient, small disturbances in the water column can obscure the stratification. Thus, small amounts of mixing during low-flow sampling can result in large variations in VOC concentrations from pumped samples.

In addition, the potential for stratification is an important consideration when selecting a sampling depth. For example, the data indicate that if the dedicated bladder pump at well MW-12 had been set about 3 ft deeper, the pump would have been in contact with water containing approximately 6,000 $\mu\text{g/L}$ more TCE than was present at the original sampling depth. If the dedicated bladder pump at well MW-5 had been set about 3 ft shallower, the pump would have been in contact with water containing approximately 690 $\mu\text{g/L}$ higher concentrations of TCE. This consideration is even more important for diffusion samplers, which sample only the water in the immediate vicinity of the sampler. Therefore, when using diffusion samplers in a well where chemical stratification is suspected within the screened interval, multiple diffusion samplers can

be used to at least initially delineate the stratification. Analytical costs during such an investigation can be minimized by using field gas chromatography to delineate the stratification and to select particular samples for more detailed laboratory analyses.

SUMMARY

The ground-water VOC concentrations obtained by using water-filled polyethylene diffusion samplers were compared to the ground-water VOC concentrations obtained by using low-flow sampling methods with a peristaltic pump and dedicated bladder pumps in observation wells at Naval Air Station North Island, California. Comparisons of VOC concentrations obtained by using bladder pumps and diffusion samplers showed a generally good correlation. Concentrations of 1,1-dichloroethene (1,1-DCE) and trichloroethene (TCE) in ground water obtained from well MW-9 obtained using the diffusion sampler agreed well (12 and 3 percent difference, respectively) with those samples obtained using the bladder pump. At well MW-5, the TCE concentration in water from the diffusion sampler was higher than in water from the bladder pump, implying that the sample collected by the bladder pump may have underestimated actual concentrations as a result of mixing. Similarly, the

higher concentrations of vinyl chloride, toluene, and total xylenes in water from the diffusion samplers in wells MW-13A and MW-13B compared to water from the bladder pump imply that the concentrations obtained by the bladder pump may have underestimated actual concentration as a result of mixing in these wells. Concentration differences between the diffusion sampling and bladder-pump sampling methods were noted in samples from well MW-12, and probably are related to mixing in a chemically stratified part of the screened interval. The findings of this investigation suggest that diffusion samplers provide a viable sampling alternative for VOCs in ground water in most tested wells at NAS North Island.

Comparisons of volatile organic compound (VOC) concentrations in water obtained by using diffusion samplers to concentrations obtained by low-flow sampling using a peristaltic pump were used to gain information on the vertical distribution of contamination in the wells. In several wells, the probable effects of mixing or volatilization during pumping resulted in lower VOC concentrations in water obtained by using the peristaltic pump compared to concentrations obtained by using the diffusion samplers; however, the data from the low-flow sampling supported the vertical VOC stratification identified by using the diffusion samplers.

Substantial VOC stratification was observed in the screened intervals of several observation wells (MW-12, MW-5, PW-15, and PW-66). The diffusion-sampler data show that the vertical change in TCE concentrations over a distance of about 5 ft was approximately 17,500 µg/L in well PW-66, approximately 7,300 µg/L in well PW-15, and approximately 5,900 µg/L in well MW-12. At well MW-5, the 1,1-DCE concentration changed by 3,410 µg/L, and the TCE concentration changed by 1,145 µg/L over a vertical distance of about 7 ft. Concentrations decreased with depth at some wells (PW-66 and MW-5) and increased with depth at others (MW-12 and PW-15). The presence of stratification in well screens is important for ground-water sampling because small disturbances in the water column can mix the stratification, resulting in large variations in VOC concentrations from pumped samples. The data imply that care must be exercised when selecting a sampling depth. When using diffusion samplers in a well where chemical stratification is suspected within the screened interval, multiple diffusion samplers can be used to at least initially delineate the stratification.

Analytical costs during such an investigation can be minimized by using field gas chromatography or indicator-tube technology to delineate the stratification and to select particular samples for more detailed laboratory analyses.

The diffusion samplers deployed in buckets containing free-phase JP-5 and Stoddard solution collected from observation wells did not show evidence of structural integrity loss during the 2 months of equilibration. The VOCs detected in the free-phase fuel also were detected in water from the diffusion samplers.

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