ERDC/CRREL TR-08-25

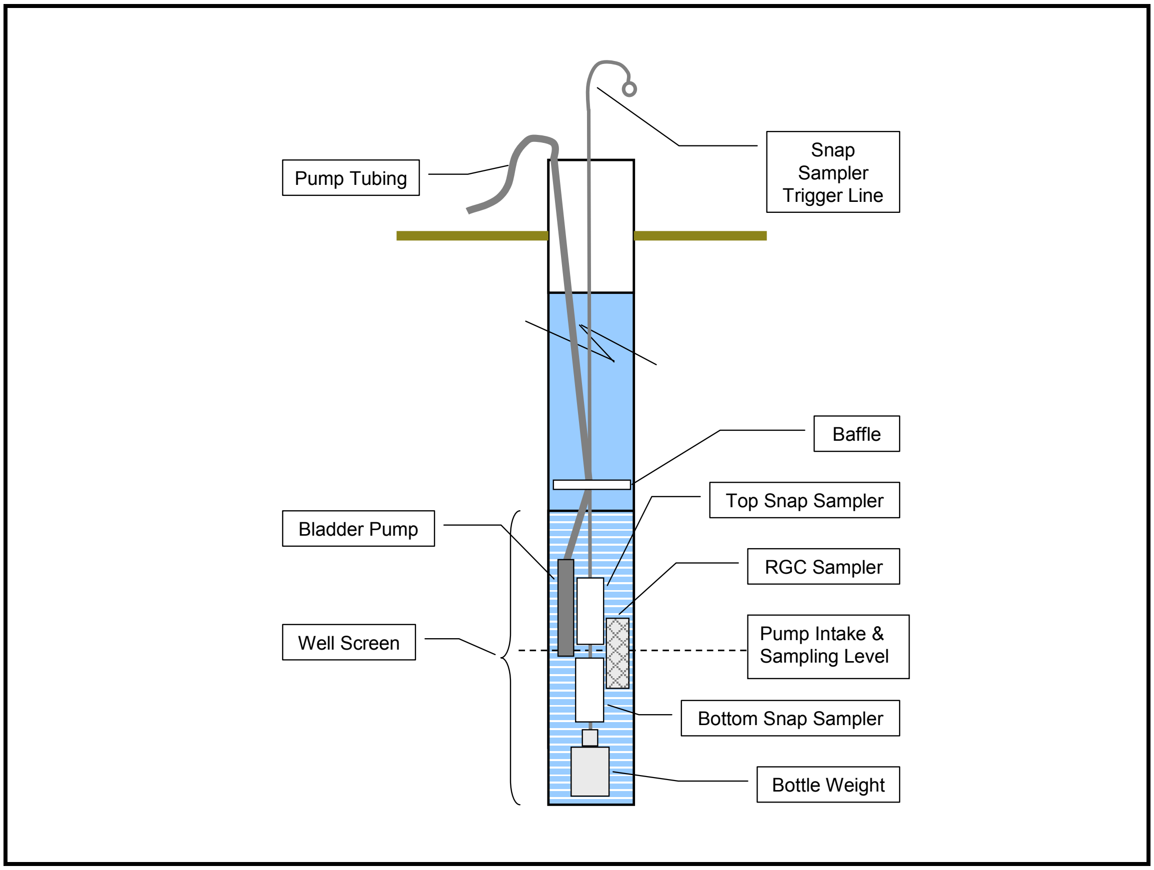
Cold Regions Research
and Engineering Laboratory

US Army Corps
of Engineers®
Engineer Research and
Development Center

Evaluation of the Snap Sampler for Sampling Ground Water Monitoring Wells for Inorganic Analytes

Louise V. Parker, Nathan D. Mulherin, and Gordon E. Gooch

December 2008



COVER: Diagram of equipment in well, including baffle, bladder pump, Snap Samplers, regenerated cellulose (RGC) sampler, and bottom weight.

ERDC/CRREL TR-08-25
December 2008

Evaluation of the Snap Sampler for Sampling Ground Water Monitoring Wells for Inorganic Analytes

Louise V. Parker, Nathan D. Mulherin, and Gordon E. Gooch

Cold Regions Research and Engineering Laboratory
U.S. Army Engineer Research and Development Center
72 Lyme Road
Hanover, NH 03755-1290

Prepared for Headquarters, U.S. Army Corps of Engineers
Washington, DC 20314-1000

Under Environmental Security Technology Certification Program ER-0630

Abstract: Laboratory studies and a field demonstration were conducted to determine the ability of the Snap Sampler to recover representative concentrations of inorganic analytes, including metals, from ground water. For the laboratory studies, concentrations of analytes in Snap Sampler samples were compared with concentrations of the analytes in samples collected from a standpipe (i.e., control samples). These studies clearly demonstrated that Snap Sampler concentrations were comparable to control sample concentrations.

For the field demonstration, 10 sampling events were conducted at the former Pease Air Force Base. Samples taken using a Snap Sampler were compared with samples collected using conventional low-flow purging and sampling and a regenerated cellulose passive diffusion sampler. Analytes included calcium, iron, magnesium, manganese, potassium, sodium, and arsenic. With one exception, Snap Sampler concentrations were equivalent to those in low-flow samples. Concentrations of unfiltered iron were higher in the Snap Sampler, especially in the two stainless steel wells, and reflected turbidity levels in the wells at the time of sampling. Elevated turbidities may have resulted from installing additional sampling equipment (including the baffle, pump, samplers, and bottom weight) in the well before sampling. We will examine this issue further at our next test site.

Contents

Report Documentation Page

Figures and Tables

Figures

Tables

Preface

We thank the Environmental Security Technology Certification Program (ESTCP) for providing the funding for this research and Dr. Andrea Leeson, Environmental Restoration Program Manager, and Dr. Jeffrey Marqusee, Executive Director, for their support.

A number of people assisted with this project, and we are most grateful for their help. We especially thank the following: Michael Daly of the U.S. Environmental Protection Agency (EPA)–Region 1 for historical field site and monitoring well data; Scott Kelley and Jenn Lane at Eastern Analytical Inc. for chemical analyses of the Pease Air Force Base (AFB) samples and very helpful advice concerning sample handling; Laura Percifield and Dr. Prem Arora, Engineer Research and Development Center-Environmental Laboratory-Environmental Processes and Engineering Division-Environmental Chemistry Laboratory (ERDC-EL-EP-EC) for handling and analyses of laboratory samples; Stephen Hastings at the U.S. Army Cold Regions Research and Engineering Laboratory (CRREL) for assisting with timely purchase and rental of field equipment and supplies; Martin Mistretta and Jim Bryant of URS Corporation at Pease International Tradeport for access, scheduling, and general assistance at the field site; and Robert Strainge, Environmental Coordinator with the U.S. Air Force Real Property Agency (USAFRPA). We give special thanks to Sandy Britt (Pro-Hydro Inc.) for his assistance and guidance throughout all the stages of this demonstration and especially for help with developing the wells and for conducting and interpreting the vertical flow testing.

Thanks to our team members, including William Major with the Naval Facilities Engineering Service Center, Dr. Jacob Gibs and Thomas Imbrigiotta with the New Jersey District of the U.S. Geological Survey, Richard Willey with the EPA–Region 1, and Don Gronstal with the USAFRPA, for their help in developing this project. Special thanks also go to our reviewers for their very useful comments and insight: Dick Willey, Tom Imbrigiotta, and Hugh Rieck with the U.S. Army Corps of Engineers Environment and Munitions Center of Expertise (EM-CX).

This publication reflects the personal views of the authors and does not suggest or reflect the policy, practices, programs, or doctrine of the U.S.

Army or Government of the United States. The contents of this report are not to be used for advertising or promotional purposes. Citation of brand names does not constitute an official endorsement or approval of the use of such commercial products.

This report was prepared under the general supervision of Dr. Terrence M. Sobecki, Chief, Biogeochemical Sciences Branch; Dr. Justin B. Berman, Chief, Research and Engineering Division, CRREL; Dr. Lance D. Hansen, Deputy Director, CRREL; and Dr. Robert E. Davis, Director, CRREL.

The Commander and Executive Director of ERDC is COL Gary E. Johnston. The Director is Dr. James R. Houston.

Unit Conversion Factors

Multiply	By	To Obtain
degrees Fahrenheit	(F-32)/1.8	degrees Celsius
feet	0.3048	meters
feet per minute	30.48	centimeters per minute
gallons	3.78541178	liters
gallons per minute	3 785 411.78	milliliters per minute
inches	2.54	centimeters
micrograms	0.001	milligrams
microns	1.0 E-06	meters

1 Introduction

It is generally accepted within the environmental practitioner community that low-flow purging and sampling methods (e.g., Puls and Barcelona 1996; U.S. EPA Region I 1996) are an improvement over older sampling methods, such as volume-based purging and sampling, and sampling with a bailer. One of the major improvements associated with this sampling method is that turbulence during sampling is reduced and this results in lower turbidity samples. Because metals and more hydrophobic contaminants (e.g., polychlorinated biphenyls and polycyclic aromatic hydrocarbons [PAHs]) can be associated with particulates and colloids (forming turbidity), low-flow sampling can reduce artificially elevated concentrations for these constituents.

However, because low-flow sampling relies on pumping, it pulls water from the most permeable zone(s) in the formation and this may mask or understate contaminant contributions from lower permeability zones. Pumping can also cause extensive mixing within the well, which can create an artificially elevated turbidity and eliminate in-well stratification that sometimes is present and may be important. Low-flow sampling can also be more expensive than passive sampling because: 1) it is time consuming; 2) dedicated sampling equipment is used or, alternatively, nondedicated pumps must be decontaminated between sampling events; and 3) it generates purge water that may require special handling and disposal. Given these issues, finding a sampling method that is less labor-intensive and costly but able to yield quality data is clearly desirable.

Recently, passive sampling techniques, such as the use of the polyethylene passive diffusion bag (PDB) sampler (Vroblesky 2001), have gained acceptance in the regulatory community (e.g., ITRC 2004, 2006, 2007; New Jersey Department of Environmental Protection 2005). Passive sampling techniques rely on the continuous natural flow through the well screen, as does low-flow sampling (Robin and Gillham 1987; Powell and Puls 1993). Most research to date (Michalski 1989; Gillham et al. 1985; Robin and Gillham 1987; Powell and Puls 1993) indicates that water in the screened portion of the well is representative of the formation if the well has been designed and developed properly. Therefore, where the use of passive sampling is appropriate to meet data quality objectives (DQOs) of the sampling effort, cost reductions can include reduced volumes of purge water waste, reduced labor during sampling, and reduced equipment costs. Based upon a field study conducted at 14 Department of Defense (DoD) sites, Parsons Inc. (2003) estimated that the potential cost savings of using passive techniques for long-term monitoring at DoD facilities could be on the order of 70% when compared with low-flow sampling. Furthermore, there can be an increased understanding of contaminant distribution with depth using passive samplers (Vroblesky and Peters 2000; Vroblesky and Petkewich 2000; Vroblesky et al. 2003; Britt 2006). To gain a more accurate picture of vertical variations in contaminant concentrations in the well, baffles or other devices may be needed to isolate the individual sampling devices.

In 2006, the Diffusion/Passive Samplers team of the Interstate Technology Regulatory Council (ITRC 2006) published an overview document on 12 passive sampling technologies. These technologies ranged in their maturity from those that were in early stages of development to others that were commercially available. The team classified these technologies on the basis of the sampler mechanism and the nature of the collected sample and determined that there were three classes of passive samplers: diffusion samplers, accumulation samplers, and equilibrated-grab samplers.

Diffusion samplers rely on diffusion of the analytes through a membrane and equilibration of analyte concentrations inside the sampler with analyte concentrations in the well water. Initially, these samplers are filled with deionized or distilled water, and then placed in the screened or open interval of a well to equilibrate. Two examples of a diffusion sampler are the PDB sampler and the regenerated cellulose (RGC or dialysis membrane) sampler. Samples collected with diffusion samplers are time-weighted toward conditions at the time the sample is collected rather than the entire period of sampler deployment. This is because the time it takes for the sampler to reflect a change in the concentration of an analyte in the well depends primarily on the rate of diffusion of each analyte through the membrane. The rate of diffusion through the membrane, described by Fick's Law, depends on the concentration gradient across the membrane, the rate of exchange of water within the well, the water temperature, properties of the membrane, and chemical and physical properties of the specific analyte. Because the pore sizes of these membranes are small (10 to 18 Å), typically only dissolved constituents are sampled with these devices.

Accumulation samplers rely on both diffusion through a membrane and then sorption by some type of sorbent material that is contained in the sampler. Samples reflect the total mass of analytes sorbed from the well water over the deployment period. The best known of these samplers is the Gore Sorber Module.

The third class of passive samplers is the equilibrated-grab samplers; examples include the Snap Sampler and the HydraSleeve sampler. These samplers are left in the well for an equilibration period before collecting a sample. This equilibration period allows time for the well to recover from any disturbance caused by placing the device in the well, for the natural flow pattern in the well to be reestablished, and for the materials in the sampler to equilibrate with the analytes in the well water thereby preventing losses of analytes due to sorption by the sampler materials. By allowing time for the well to recover before collecting the sample, the well is less agitated during the sampling event and particles that are not normally mobile in the formation are less likely to be entrained in the sample when it is collected. The time needed for equilibration will depend upon the sampling device (and materials in the sampler), the physical and chemical properties of the analyte(s), the ambient flushing rate of the well, and water temperature. The ITRC Diffusion/Passive Samplers team (ITRC 2006, 2007) recommends a minimum deployment time of two weeks for these and most other passive samplers.

Advantages associated with using equilibrated-grab samplers

Although most passive samplers are easy to use and offer substantial cost savings, many are limited with respect to the types of analytes that they can be used to sample for. As an example, the PDB sampler can only be used for volatile organic compounds (VOCs), and the Gore Module can only be used for VOCs and semivolatile organic compounds (SVOCs). Although the RGC sampler can be used for a broader range of analyte types including inorganic analytes, the deployment time for the RGC sampler may be limited because the membrane is subject to biodegradation. In contrast, equilibrated-grab samplers recover a whole-water sample (versus dissolved constituents) and thus potentially can be used to collect samples for a broad range of analyte types. Furthermore, unlike the other two classes of passive samplers, these samplers are able to collect a sample that is for all practical purposes ‘instantaneous’ in time (rather than a time-weighted average).

Two features that distinguish the Snap Sampler from other passive samplers are that the sample is collected in sample bottles sealed under in-situ conditions and the sample does not have to be transferred to another container at the surface. These features eliminate any possible interaction of the sample with the water column as the device is brought to the surface; they also reduce losses of volatiles or changes in dissolved gases that can occur at the well head during bottle filling and affect concentrations of metals subject to oxidation/precipitation reactions.

In addition, because the Snap Sampler is not triggered until after the well has recovered from any disturbance caused by placing the sampler in the well, presumably samples should not have artificially elevated turbidity but would contain any naturally mobile colloid-borne contaminants. In contrast, even low-flow sampling has been shown to artificially elevate particle levels in some wells and hydrologic settings (Bailey et al. 2005); Puls and Paul (1997) found that in one of the wells they sampled using the Diffusion Multilevel Sampler (developed by Ronen et al. [1987]) colloidal particles were entrained in the sampler cells with time, thereby yielding falsely elevated chromium (Cr) concentrations in these samplers. (Apparently the particles entered through the porous membrane and settled out before they could exit the membrane.)

Description of the Snap Sampler

The Snap Sampler consists of four components: a bottle that has openings on both ends and contains spring-loaded end caps, a sampler body that holds the bottle and contains the trigger mechanism that closes the bottles, a trigger line, and a docking station (see Figure 1). The spring in the bottles is coated with perfluoroalkoxy (PFA) (Teflon) and is connected to PFA end caps at both ends of the bottle. Currently, 40-mL glass volatile organic analyte (VOA) vials and 125-mL high-density polyethylene (HDPE) bottles that fit in a 2-inch-diameter well (or larger) and 350-mL HDPE bottles that fit in a 4-inch-diameter well (or larger) can be obtained from the developer/manufacturer (ProHydro Inc.; http://www.snapsampler.com/). (Earlier 125-mL bottles were made of glass and polypropylene [PP].) The Snap Sampler body is made of either stainless steel or acetal (Delrin) plastic and holds the bottles in the open position with a release-pin system. The trigger line connects to the release pin and consists of a movable internal fluorinated ethylene polypropylene (FEP)-coated (Teflon) stainless steel cable surrounded by plastic tubing. Depend-

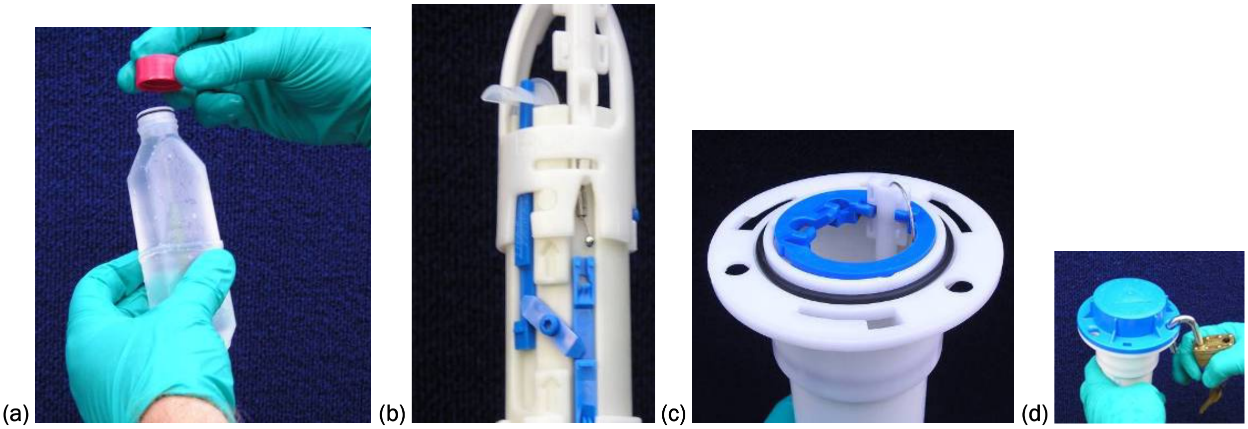


Figure 1. Snap Sampler (a) 125-mL PP bottle, (b) Snap Sampler body, (c) trigger line attached to docking station, and (d) docking station with locking cap.

ing on the user's sampling needs, two types of plastic tubing are available, HDPE tubing and polyvinylidene fluoride (PVDF) (Kynar) tubing. Since this study was conducted, an electric trigger mechanism has been developed. This device uses a down-hole actuator and the samplers are activated to close with a switch that is at the surface.

The samplers are deployed in the well with the end caps of the bottle(s) in an open position. After the equilibration period is complete, the trigger is activated to close the sample bottle(s). Once retrieved, the sample can remain in the sampler bottle. These samplers fit in 2-inch diameter and larger wells. To obtain a larger sample volume or more than one sample at a time, two to four samplers can be deployed in series on the same trigger line or on separate trigger lines (Figure 2). (More information on how to deploy a Snap Sampler can be found in Parker and Mulherin [2007] or on the manufacturer's Web site.)

Previous research studies

Parsons Inc. (2005) conducted a field test at the former McClellan Air Force Base that evaluated six different passive samplers. This field test was conducted for the Omaha District of the U.S. Army Corps of Engineers, Air Force Center for Environmental Excellence, and Air Force Real Property Agency. In this study, analyte concentrations in samples taken with the various passive samplers were compared with analyte concentrations in samples that were taken using two pumped sampling methods. The two pumped methods included a low-flow purging and sampling method and a well-volume purging and sampling method (where the well was purged of 3 to 5 well volumes and then a sample was collected using a bailer). The six passive samplers included several diffusion samplers (including the PDB sampler and RGC sampler) and two equilibrated-grab devices including the Snap Sampler. For the Snap Sampler, analytes that were compared included several VOCs (including 1,4-dioxane) and anions. This work has not been published in a peer-reviewed journal but the final report is available online (see Parsons [2005] for the link).

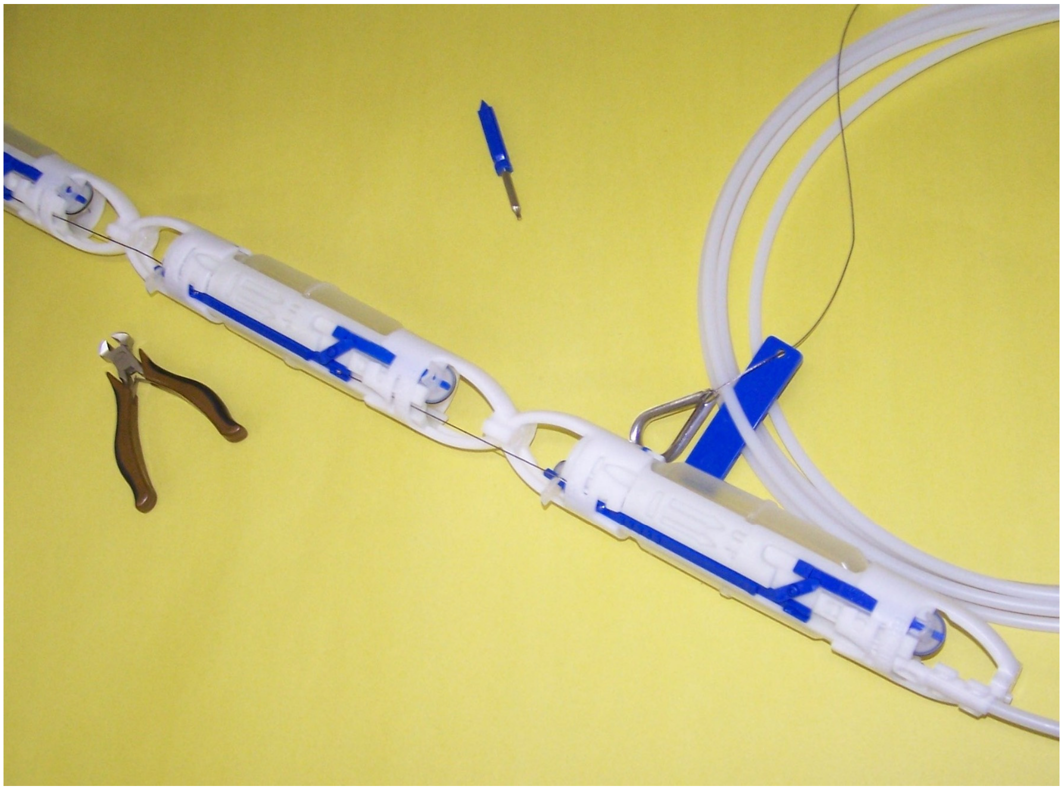


Figure 2. Assembled string of two Snap Samplers attached to the trigger line.

Although a large data set was generated in this study, interpreting the results is difficult. This is due in part to differences in sampling (samples were taken on different days), sample handling methodologies (e.g., some samples were poured into a second container while others were not), the numerous methods that were used to analyze the data, and issues with some of the statistical analyses (e.g., the data were pooled for the various analytes before conducting the statistical analyses).

However, regression plots of the pooled VOC data were informative. These plots showed that the Snap Sampler VOC concentrations correlated well with those taken using low-flow purging and sampling ($r^2 = 0.995$) although, the Snap Sampler concentrations were higher (the slope was 1.77). In contrast, a similar comparison of the Snap Sampler and 3-well volume samples indicated that the concentrations of VOCs agreed well ($r^2 = 0.90$ and a slope of 1.04). The findings were similar for the anions. That is, Snap Sampler concentrations of the anions were higher than those in the low-flow samples (with a slope of 1.22) but were similar in magnitude to those in the 3-well volume purged samples (with a slope of 1.08). Parsons concluded that "the 'high bias' in the Snap Sampler concentrations suggested that they may be more representative of the actual concentrations in the well at the time of sample collection, particularly for VOCs." However, they also noted that another reason concentrations of volatiles were higher in the Snap Sampler samples was because there was less transfer and loss associated with these samples than the other samples that were collected in two 20-mL vials and then transferred to a 40-mL vial (in the laboratory). However, they did not offer any explanation as to why the anion concentrations would also be higher with this sampler. Given that the findings were similar for the anions, this suggests to us that these differences were because the water sampled was not the same, i.e., the water quality of the low-flow purging and sampling samples was different from that in the Snap Sampler. In contrast, the purged-volume sampling method collected water that was similar in character to that sampled with the Snap Sampler. This may be an important distinction that merits further study.

Our laboratory has conducted a series of studies that evaluated the ability of the Snap Sampler to recover representative concentrations of VOCs and explosives in ground water (Parker and Mulherin 2007). These studies include both laboratory studies and field studies. The laboratory studies were conducted at room temperature (~20° to 22° C) in a 244-cm (8-ft) tall, 20-cm (8-inch) diameter PVC standpipe that contained known concentrations of either a suite of VOCs or explosives. Statistical analyses of the data (on an analyte-by-analyte basis) were conducted to determine if the concentrations of analytes in samples taken with the Snap Sampler were significantly different from known concentrations of the analytes in samples collected from the standpipe (i.e., control samples) (at the 95% confidence level, $\alpha = 0.05$). Analysis of these data revealed that the Snap Sampler recovered comparable concentrations of VOCs and explosives after a brief equilibration period. For VOCs, approximately 3 days was needed for the analytes to reach equilibrium with the materials in the sampler (i.e., for there to be no losses due to sorption by the materials), whereas for explosives, approximately 24 hr was sufficient. Clearly, the minimum 2-week equilibration time[1] recommended by the ITRC Diffusion/Passive Samplers team for most passive samplers (ITRC 2007) allows more than enough time for the analytes to equilibrate with the materials in this sampler, even if the ground water temperature was cooler than the 20°C used in these studies.

Parker and Mulherin (2007) also conducted three field studies to evaluate the ability of this sampler to recover equivalent concentrations of VOCs and explosives when compared with the EPA's low-flow purging and sampling (U.S. EPA Region I 1996). Sites included our own laboratory grounds (at the U.S. Army Corps of Engineer's Cold Regions Research and Engineering Laboratory [CRREL]) in Hanover, NH; the Silresim Superfund site in Lowell, MA; and the former Louisiana Army Ammunition Plant (LAAP) in Minden, LA. At the CRREL test site, one of our trichloroethylene (TCE)-contaminated wells was sampled on five different days using both sampling methods. At the Silresim site, four wells were sampled for 13 VOCs that included the aromatic hydrocarbons common in gasoline and other fuels (benzene, toluene, ethylbenzene, and the xylenes [BTEX compounds]) and several chlorinated VOCs. At LAAP, five wells were sampled for seven explosive compounds and their daughter products. For these studies, concentrations of analytes in samples taken using the Snap Sampler were compared with concentrations of the analytes in samples taken using the EPA's low-flow purging and sampling protocol. Again statistical analyses were used to determine if there was a statistically significant difference (at the 95% confidence level, $\alpha = 0.05$) between the individual analyte concentrations in these samples.

The results of the statistical analyses of the data from the CRREL site and the data from the Silresim site indicated that there were no statistically significant differences between the VOC concentrations in the samples taken with the Snap Samplers versus those in the samples taken using low-flow purging and sampling. A similar comparison of the data from the former LAAP also indicated that the Snap Sampler recovered concentrations of explosives comparable to those obtained by low-flow sampling. However, we did encounter some equipment problems with the Snap Sampler when we tried to deploy three of the larger (125-mL) glass bottles in tandem at this site. In contrast, at the CRREL and Silresim sites where we had deployed only one or two VOA vials on the trigger lines, we did not

---

[1] The intent of the ITRC 2-week equilibration time is to allow the well to recover from any disturbance caused by placing the sampler in the well.

encounter any problems with the equipment. Since then, the developer has made a number of design improvements in this sampler technology and has replaced the 125-mL glass bottles with 125-mL plastic bottles that are lighter in weight and have a better spring. We tested the newer plastic bottles in one of our deeper (135-ft) wells and found that we were able to trigger three bottles in tandem on a single line successfully. However, when we deployed four samplers on a single line (the maximum number of samplers recommended per trigger line by the manufacturer), we found that we had to be certain that there was no tension on the connector cables to each of the bottles. This means that in deeper wells, it may be advisable to use either an additional trigger line to deploy four samplers in the well or use an electric trigger. The additional cost of an additional trigger line for a long-term monitoring program would be insignificant.

We concluded that this sampler is a promising cost-effective technology for sampling ground water monitoring wells for explosives and VOCs under in-situ conditions.

Using passive samplers to profile contamination with depth in wells

One interesting application of passive sampling is to profile contamination in the well with depth. This capability is based upon the observation that (under ambient flow conditions) stratification of contaminants is observed within the screened interval of some wells, although the relationship between in-screen stratification and aquifer stratification cannot always be determined. In other wells, stratification is eliminated by diffusive mixing or by hydraulically driven mixing due to vertical flow in the well, or is not present due to absence of stratification in the aquifer. Vertical flow results when the well screen intersects zones of different hydraulic head, a condition that is more likely to be found in wells with longer screens (e.g., those that are 20 ft in length and longer) (Elci et al. 2001). Convective circulation within wells can also result when there is an unstable density gradient in the well water. A density gradient can result from differences in the concentration of constituents in the water or from differences in temperature of the water within the well (Diment 1967; Gretener 1967; Sammel 1968; Britt 2005; Vroblesky et al. 2006). In instances where there is convective flow, oxygen can be transported from shallower parts of the well to deeper parts of the well, including the screened portion of the well (Vroblesky et al. 2006). This has been shown to affect not only the ability of a passive sampler to recover representative samples, but also to affect the ability of low-flow purging and sampling to recover representative samples. However, Vroblesky et al. (2006) demonstrated that either an in-well inflatable packer or a (polyethylene disc) baffle could be used to prevent temperature-induced convective flow in wells (when placed at the top of the well screen). This approach worked equally well for both passive and low-flow sampling methods.

Baffles have been used in other studies to inhibit in-well mixing. As an example, when Britt (2006) used baffles and Snap Samplers to profile phenol contamination in a well, he found a difference in concentration of five orders of magnitude along the 10-ft length screen. In contrast, previous samples taken using the Snap Sampler with no baffles in the well indicated that there was little difference in the contaminant concentrations with depth.

Although using mixing inhibitors in wells substantially reduces mixing in the well, the impact they would have on any mixing that may occur in the filter pack is not known. However, using mixing inhibitors in wells appears to be a promising approach for better delineating stratification in a formation using existing monitoring wells.

2 Purpose

The purpose of these studies was to determine whether the Snap Sampler was able to recover representative concentrations of inorganic analytes, especially metals. Laboratory studies were used to demonstrate whether this sampler could recover representative concentrations of cations and anions when compared with known concentrations of these analytes, and a field demonstration was conducted to determine whether this sampler was able to recover samples with equivalent concentrations of metals to those obtained in samples taken using the EPA's low-flow purging and sampling protocol.

3 Materials and Methods

Before conducting a field demonstration of the Snap Sampler, the ability of the Snap Sampler to recover representative concentrations of inorganic analytes, including metals, was tested in the laboratory. These studies were conducted using a standpipe that contained known concentrations of analytes. Control samples taken from a spigot on the standpipe were compared with samples taken using Snap Samplers. For each analyte, statistical analyses were used to determine if the concentrations of analytes in the samples taken with the Snap Sampler were significantly different from the control samples.

The field demonstration was conducted at the former Pease Air Force Base (AFB). Inorganic analytes with measurable concentrations included arsenic (As), calcium (Ca), iron (Fe), magnesium (Mg), manganese (Mn), potassium (K), and sodium (Na). Analyte concentrations in samples collected using the Snap Samplers were compared with concentrations of these analytes in samples collected from the same well and sampling depth using low-flow purging and sampling. Again for each analyte, statistical analyses were used to determine if there was a statistically significant difference between concentrations in the Snap Sampler samples versus those in the low-flow samples. Both filtered and unfiltered samples were compared. In addition, because passive diffusion samplers typically exclude particulate matter, additional samples were collected using a passive diffusion sampler, the RGC sampler. Including this sampler in the study allows us to compare analyte concentrations in filtered and unfiltered low-flow samples and Snap Sampler samplers with (naturally filtered) RGC samples, and this gives us a measure of what analytes were naturally mobile as colloids at this site.

Laboratory studies

The 8-inch-diameter, 8-ft-tall PVC standpipe used in these studies is the same as described previously (Parker and Mulherin 2007). It was filled with the test solution to within ~10–13 cm (4–5 inch) of the top. This allowed enough room to place the samplers in the standpipe without having the test solution overflow the standpipe. The top of the standpipe was covered with a thin plastic disc that was then covered with aluminum foil. These studies were conducted at room temperature (~20°C to 22°C).

In all these studies, the samplers were deployed at the same depth as the sampling port (spigot) on the standpipe (80 cm from the bottom of the standpipe). Typically, paired samples were obtained by either collecting a sample with the Snap Sampler and then drawing a control sample from the spigot on the standpipe, or vice versa. The order in which the two samples were obtained was alternated until all the sample-control pairs were collected, so that any bias due to sampling order was eliminated.

First standpipe study

The test solution contained six metals: arsenic (As), cadmium (Cd), chromium (Cr), lead (Pb), nickel (Ni), and zinc (Zn). Concentrations of As, Cd, Cr, and Pb were ~200 ppb, and concentrations of Ni and Zn were ~400 ppb. The test solution was prepared by adding known volumes of As, Cd, Cr, Ni, and Zn atomic-adsorption reference solutions to a 1000-mL glass beaker containing deionized (DI) water. A known weight of lead nitrate was then added to the test solution and the solution was stirred for ~2 hr. The concentrated test solution was added to the standpipe, and the diluted test solution was then circulated for ~2½ hr to mix it. This was accomplished by placing a hose in the bottom of the standpipe and pumping the test solution to the top of the standpipe using a peristaltic pump.

Five Snap Sampler trigger lines were deployed in the standpipe, and there were 125-mL PP Snap Sampler bottles on each of the trigger lines. The trigger lines were deployed such that the middle bottle was at the same depth as the spigot on the standpipe. After 24 hr, the first trigger line was triggered. Three control samples were collected from the spigot on the standpipe in 40-mL pre-cleaned VOA vials. The Snap Sampler trigger line (with the three bottles) was then removed from the standpipe, and the bottles were removed from the Snap Sampler body and capped (without removing the springs from the bottles). The samples were preserved by adding ~0.32 mL of 70% Ultrex nitric acid to the control samples and ~1.0 mL of the acid to the Snap Sampler bottles. The samples were then placed on ice and sent by overnight courier to U.S. Army Engineer Research and Development Center-Environmental Laboratory-Environmental Processes and Engineering Division (ERDC-EL-EP). A similar protocol was followed after 48- and 72-hr equilibration.

Analyses were by inductively coupled plasma (ICP) using EPA SW 846 method 6010B (U.S. EPA 1996).

After receiving the results from these analyses, we collected a fourth set of samples (on the fourth trigger line) after 50 days and a final set of samples after 102 days. The protocols used were the same as described previously.

Data analyses

The percent recovery of the Snap Sampler compared with the recovery for the control samples was determined for each sampling day and analyte. In addition, the percent relative standard deviation (% RSD) was determined for each type of sample (i.e., Snap Sampler or control sample), sampling day, and analyte.

The data were also analyzed using standard statistical tests to determine if there were statistically significant differences between the analyte concentrations in the Snap Sampler samples versus those in the control samples. The data for each analyte and sampling day were analyzed independently. For all data sets where the data were normally distributed and the variances were homogeneous, a standard *t*-test (at 95% confidence level, $\alpha = 0.05$) was used. In instances where the data were not found to be normally distributed (using a *P* value of 0.050), the data were first log transformed and then tested for a normal distribution and homogeneity of the variances. In those instances where these criteria were met, a *t*-test was then performed on the log-transformed data. In instances where neither the raw data nor the log-transformed data met these criteria, a Mann-Whitney rank sum test was used to compare the data.

Second standpipe study

For the second laboratory study, the metal spigot on the standpipe was replaced with an HDPE spigot. The test solution was prepared by adding calcium nitrate, manganese sulfate (monohydrate), sodium chloride, potassium bromide, and magnesium perchlorate to a 5-gallon glass carboy containing DI water. The carboys were filled so there was minimal headspace and then covered with two layers of tightly fitting aluminum foil. To dissolve the analytes, the solutions were stirred for ~1 hr using a magnetic stirrer (with a Teflon-coated stir bar). After stirring the test solution, the carboy was added to the standpipe that already contained ~4 gallons of DI water. The diluted test solution was then mixed by emptying the standpipe and refilling it. Concentrations of the cations in the test solution ranged from ~8 mg/L (for Mg) to ~42 mg/L (for K). Concentrations of the anions in the test solution ranged from ~13 mg/L (for nitrate) to ~80 mg/L (for bromide).

The experimental protocol used was similar to that used in the first laboratory study except that only three Snap Sampler trigger lines were deployed in the standpipe and each line contained only two 125-mL HDPE Snap Sampler bottles. One trigger line (with two bottles) was recovered after 24-, 48-, and 72-hr equilibration. Samples were acidified and sent to ERDC-EL-EP for analyses by ion chromatography using EPA methods 300.0 (Hautman et al. 1999) and 314.0 for perchlorate (Pfaff 1993).

Data analyses

The percent recovery of the Snap Sampler compared with the recovery for the control samples was determined for each sampling day and analyte. The % RSD was also determined for each type of sample (i.e., Snap Sampler or control sample), sampling day, and analyte. There were not enough replicate samples to allow us to perform statistical analyses of the data from this study.

Field demonstration

Site description

The former Pease AFB is located in the southeastern portion of New Hampshire in the town of Newington and city of Portsmouth. The former base occupies approximately 4365 acres on a peninsula that is bounded by Great Bay, Little Bay, and the Piscataqua River. The unconsolidated stratigraphic units identified at Pease AFB are fill, Upper Sand (US), Marine Clay and Silt (MCS), Lower Sand (LS), and glacial till. One or more of these units may be absent at any particular location. The underlying bedrock is either the Kittery or Eliot formation, depending on the specific site location. More information on this site can be found in Appendix A.

Eight 10-cm- (4-inch-) diameter ground water monitoring wells were selected for this study; six of the wells were constructed with PVC risers and screens and two of the wells (32-5020 and 32-6195) were constructed with stainless steel casing and screens. The criteria used to select these wells included well diameter, screen length, sampling depth, and detectable concentrations of the analytes of interest. The tops of the well screens varied in depth from 1.5 to 13.7 m (4 to 45 ft) below ground surface (bgs), and screen lengths varied from 1.5 to 4.6 m (5 to 15 ft). The tops of all the screens were below the water table. The construction details for these wells are given in Table 1. Six of the wells were located near building 113 (site 32, zone 3), which is near a spill from an underground TCE storage tank. The other two wells were located in a former bulk fuel storage area (site 13, zone 1). These locations are shown on the base map in Figure 3.

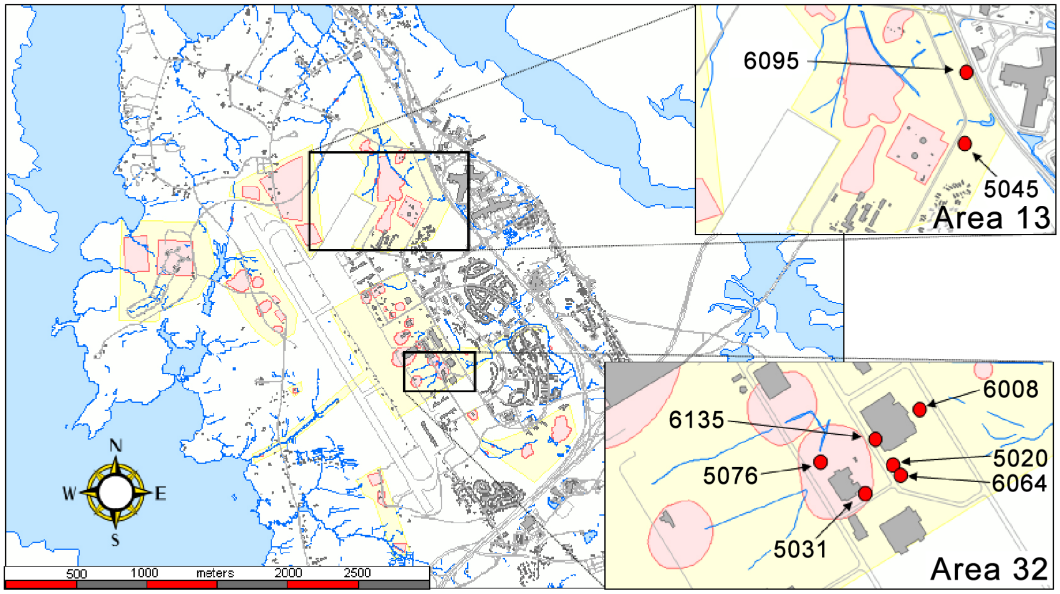


Figure 3. Map of the former Pease AFB showing the location of the wells used in this study.

Preliminary well assessment and other site activities

Well redevelopment

Preliminary assessment of the wells revealed that most of them had large amounts of silt in them. Because it was important that the wells function properly, we redeveloped all of them before conducting any further field work.

Flow-meter testing

Once the wells were redeveloped, the hydraulics in each well was determined by using a heat pulse flow meter manufactured by Mount Sopris Instruments (Golden, CO). These tests were conducted under both static and low-flow pumping conditions.

The heat pulse flow meter works by generating a “heat pulse” that is directly transferred to the water surrounding the element. The heated water

Table 1. General information on the monitoring wells used in this study.

Well No.	Well Diameter (inches)	Well Casing Material	Concentrations Above MCLs?	Top of Screen (ft bgs)	Bottom of Screen (ft bgs)	Screen Length (ft)	Approximate Depth to Water (ft bgs)	Screen Length in Water Table (ft)
Bulk fuel storage area, Site 13								
13-5045	4	PVC	Arsenic	5	13	8	3	All
13-6095*	4	PVC	Arsenic	11	21	10	6	All
Site 32								
32-5020	4	Stainless	Arsenic	14.7	25.35	10.65	5	All
32-5031	4	PVC	Arsenic	13	18	5	4	All
32-5076	4	PVC	Arsenic	35	45	10	5	All
32-6008*	4	PVC	Arsenic	38	52	14	3	All
32-6064*	4	PVC	Arsenic	34.5	44.5	10	5	All
32-6135*	4	Stainless	Arsenic	45	60	15	7	All
*Bedrock well.								

then travels vertically along with the flow in the well, if any vertical flow is present. Sensors above and below the heating element detect the arrival of the heated water, which is recorded and displayed on the field computer used to run the tests. Travel time of the heat pulse, along with borehole diameter is used to calculate flow rate within the bore. The flow-meter software reports the flow rate in gallons per minute (gpm). In the case where there is no vertical flow in the well, the heated water will slowly approach the upper sensor as a result of the buoyancy of the heated water. This artifact upward vertical flow can be discerned from true vertical flow by comparing velocities in known locations where there is no vertical flow (e.g., in the blank casing).

Calibrations to "no vertical" flow were conducted in the blank casing in the first well. The calibration was conducted to establish a baseline for appearance of the buoyant heat pulse at the upper sensor under known no-flow conditions. Repeated measurements were consistent showing an upward "flow" of less than 0.01 gpm. Arrival of the heat pulse under no-flow conditions was 70–90 sec. This instrument could not detect a flow rate lower than 0.01 gpm in these wells

Ambient vertical-flow testing

For the ambient flow testing, the flow meter was deployed at the middle of the screen. The tool was then left to equilibrate in the well for 10 to 30 min. This allowed the water level in the well to be reestablished after insertion of the probe. After the equilibration, several readings were collected at each well to determine consistency of the results.

Dynamic vertical-flow testing

After completion of the ambient testing in a well, dynamic (pumping) tests were conducted. The heat pulse flow meter was left at midscreen position during the first phase of the dynamic tests. Flow within the screen interval was induced by pumping with a peristaltic pump above the screen interval in each well (all wells had fully submerged well screens—no wells were water table wells). Low flow rates were used to simulate flow rates anticipated for subsequent low-flow purging and sampling. Flow rates at the wells during the dynamic tests ranged from 170 to 430 mL/min (0.045 to 0.11 gpm). Steady-state (no further drawdown) conditions were established before collecting measurement data. Several heat pulse measurements were taken at each midscreen deployment to determine consistency of the measurements.

After the midscreen measurements, the flow-meter sensor was raised above the level of the screen. Additional measurements were conducted in the blank casing at the same pumping flow rate. These measurements allowed a calibration of the full flow rate at each well. During steady drawdown conditions, all water being pumped must come from the well screen. Therefore, simple calculations can be made using the midscreen flow rate and full-screen flow rate to estimate contribution of the upper and lower portions of the well.

Profiling contaminant concentrations with depth in the wells

After completion of the borehole flow-meter tests, two Snap Samplers, each on its own trigger line, were deployed in each well (Figure 4). The bottom sampler had a baffle that divided the well screen in half, i.e., yielding an upper and lower zone. The upper sampler had a baffle that was placed 0.5 ft above the top of the well screen. This was done to separate

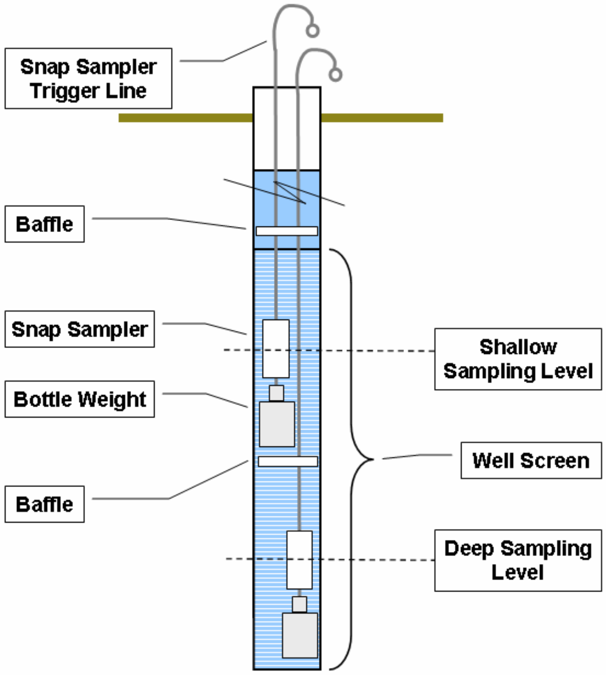


Figure 4. Diagram showing interval Snap Samplers in a well.

the screen from the blank casing. Both samplers were placed in the middle of their respective zones. The decision was made not to vertically profile well 32-5031 because the screen was only 5 ft in length so only one sampler was placed in this well. However, like the other wells, a baffle was placed above the sampler to separate the screen from the blank casing. The samplers were left in the well for 1 week before collecting the samples.

The baffles consisted of two circular discs of 0.030-inch-thick polyethylene, sized slightly larger than the inside diameter of the well. Around the outside of the disc, slits were cut to allow the baffles to bend slightly so that the baffle fit tightly within the well. The slits in the two discs were misaligned to limit water exchange. The discs are attached to the Snap Sampler trigger line with plastic snap-on grips ("Herbie" clamps). Figure 5 shows a trigger line with a baffle, spooled and ready for deployment.

Once the samples were collected, they were placed on ice, acidified, and sent by overnight courier to the laboratory for analyses.

Equipment blanks

Before deploying the pumps in the wells, equipment blanks were drawn in the laboratory. The pumps were placed in a 94-cm-tall by 16.5-cm-diameter (37-inch x 6.5-inch) pipette washer containing deionized (DI) water. All the pumps were connected to the pump controller at the same time via a manifold of Teflon tubing and left pumping overnight. During this time, the pumps were pressurized for 1 sec and exhausted for 1 sec at a gauge pressure of approximately 6–8 psi. Pumped water was drained into the sink while the pipette cleaner was continuously replenished with DI water. After 23 hr of pumping, the pumps were shut off for ~4 hr. The pipette cleaner was then emptied and refilled with fresh DI water. After another 45 min of pumping continuously replenished DI water, sample blanks were pumped through (6.35-mm-diameter and ~0.75-m-long) Teflon-lined polyethylene tubing into 125-mL HDPE bottles containing diluted nitric acid as a preservative. After all the pumps had been removed from the pipette cleaner, two water blanks were drawn from the bottom drain of the chamber. The sample blanks were stored in a refrigerator overnight, packed on ice the next morning, and were transported by courier service to the contract laboratory for analysis by EPA method 200.8 (Creed et al. 1994).

Figure 5. Snap Sampler trigger line with baffle.

Blanks were also obtained for the RGC and Snap Samplers. For the Snap Sampler blanks, two trigger lines, each containing one 125-mL plastic Snap Sampler bottle, were deployed in a pipette cleaner filled with Milli-Q water. One sample was to be filtered and one was to remain unfiltered. For the RGC sampler blanks, a RGC sampler was placed in a 24-cm-tall x 19-cm-diameter (9.5-inch x 7.5-inch) HDPE jar[1] filled with Milli-Q water. This sample would not be filtered (as the sample was already filtered by the small pore size of the membrane). Two weeks later, the Snap Samplers were triggered and removed from the pipette cleaner. The Snap Sampler bottles were removed from the trigger line, and the samples remained in the Snap Sampler bottles. One of these samples was then acidified with 0.5 mL of nitric acid and the other was to be filtered and acidified in the laboratory. Then, two water samples were collected in 125-mL HDPE bottles from the bottom drain of the pipette cleaner, and the sample that was to remain unfiltered was then acidified in a similar fashion with nitric acid. At this time, the RGC sampler was removed from its container and the sample was transferred to a 125-mL HDPE plastic bottle containing nitric acid. All of these samples were placed in a refrigerator at 4°C and left for the weekend. The following Monday, the samples were packed on ice and taken directly to the contract laboratory by a courier.

---

[1] This chamber was subsequently used to sparge the RGC samples before deploying them in the wells.

Initial low-flow sampling of the wells

Before conducting any low-flow sampling, the equipment used to monitor the purge parameters was checked approximately a month before we began the project and again just before beginning the field work.

After a small diameter (¾-inch) bladder pump (the Mini Bladder Pump), manufactured by Durham-Geo Slope Indicator (Stone Mountain, GA), was installed in each of the wells used in the field study, preliminary unfiltered low-flow samples were collected from all the wells. This was done to: 1) determine analyte concentrations, 2) confirm that the flow rate to be used would not cause excessive drawdown in the wells, 3) determine the sampling time for low-flow sampling (needed for the cost analyses), and 4) equilibrate the materials in the pump and tubing with the analytes in the well water (to reduce possible losses due to sorption that might occur).

The samples were collected using the EPA's low-flow purging and sampling protocol (Puls and Barcelona 1996; U.S. EPA Region I 1996). The purge parameters were monitored using a Horiba (MDL W-22XD) probe and a flow-through cell. These included turbidity, dissolved oxygen (DO), conductivity, salinity, pH, total dissolved solids (TDS), redox potential (ORP), and temperature. These parameters were monitored until three successive readings varied by no more than +10%. Furthermore, in most cases, samples were not collected until the turbidity reading was stable and a value of 10 nephelometric turbidity units (NTU) or less was obtained. It typically took between 30 and 90 min for the purge parameters to stabilize, although some wells took considerably longer to stabilize. The Horiba probe was calibrated each morning using the Horiba autocalibration solution. This solution is used to calibrate the meter for pH, conductivity, turbidity, and DO. (The other purge parameters are calculated based on these measurements.)

The samples were collected in 125-mL acid-cleaned HDPE bottles containing diluted nitric acid (as described previously). After the sample bottles were filled, they were placed on ice in a cooler, and then shipped to the contract laboratory for analyses.

Field demonstration

Sampler deployment

Pre-cleaned regenerated cellulose (CelluSep H1) membrane in preservative solution was purchased for this study. The molecular weight cutoff for this membrane is 8000 Da, and the average pore size is 0.0018 µm. The RGC samplers were constructed the morning before we departed for the field site, according to the construction protocol given by the ITRC passive sampler team (ITRC 2007). After construction, the samplers were placed in 1-gallon plastic, zip-closure bags filled with DI water, and placed in an ice cooler for transport to the field. On the morning of the first day of sampler deployment, four samplers were placed in a wide-mouth (7.5 inches in diameter by 9.5 inches tall) HDPE screw-cap bottle that was filled with DI water so that the samplers were fully immersed. Nitrogen gas was then fed under low pressure via tubing to the bottom of the container. The samplers were sparged for at least 90 min. When sampling was complete at that site, the tank was disconnected and a replacement RGC sampler was added to the bottle; the sparging process was then repeated at the next site. The samplers were not sparged overnight but the same process was followed each day of deployment. For the second round of sampling, four RGC samples were sparged for 2 hr in the laboratory before taking them to the field. At the site the next day, the samplers were again sparged for at least 1 hr before deployment in the well.

For the field demonstration, we placed a ¾-inch diameter bladder pump, a Snap Sampler trigger line with two Snap Samplers and a baffle, and a RGC sampler in each of the monitoring wells (Figure 6). A small-diameter pump was used in these studies so that all the equipment could be placed in the well at the same depth at the same time. The pump was placed at approximately the midpoint of the well screen. The Snap Samplers were placed in the well so that they straddled (i.e., above and below) the entry port of the bladder pump. The baffle was positioned 0.5 ft above the top of the well screen. The RGC samplers were placed at the same depth as the pump inlet. In the first round of sampling, the RGC samplers were attached to the side of the pump (so they did not cover the pump inlet) and the Snap Samplers were deployed separately. In the second round of sampling, the three pieces of sampling equipment were bundled together

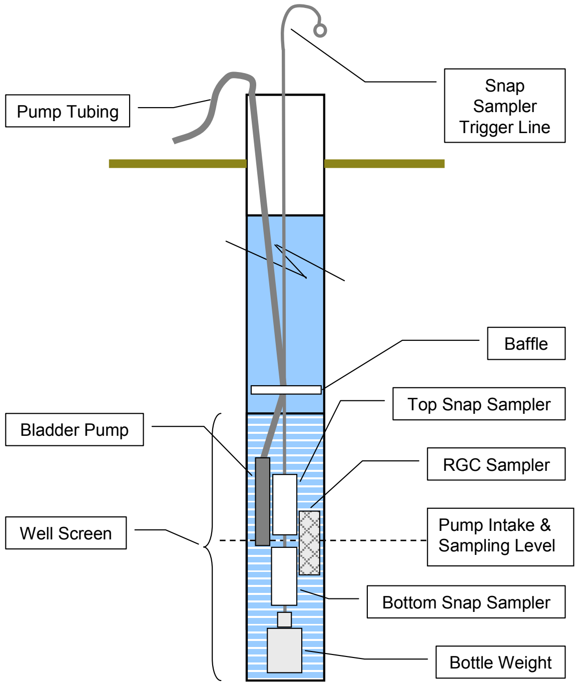


Figure 6. Diagram of equipment in well, including baffle, bladder pump, Snap Samplers, RGC sampler, and bottom weight.

(again making certain that the pump inlet was not covered with other equipment). To insure the integrity of the RGC samplers, these samplers were not constructed until the day before the field demonstration.

The equipment was then left to equilibrate for 2 weeks before collecting the samples.

Sample collection

There were two rounds of sampling conducted for this demonstration. In the first round of sampling, all eight wells were sampled. In the second round of sampling, only two wells were sampled: 32-6064 and 32-5020.

For the first two wells that were sampled in the first round of sampling (i.e., well numbers 32-6064 and 32-5020), the sampling order was as follows. The Snap Samplers were triggered to collect the sample but were left in the well. The pump with the attached RGC sampler was removed from the well, the RGC sampler was removed from the pump, and then an aliquot of the RGC sample was transferred to a 125-mL acid-cleaned HDPE bottle, containing 0.4 mL of (trace-metal grade) diluted nitric acid (diluted 1:1 with DI water), and the sample bottle was placed on ice in a cooler. The Snap Sampler line was then removed from the well, the Snap Sampler bottles were removed from the sampler bodies, the springs were removed from the bottles, the bottles were capped, the bottom sample bottle was acidified, and the bottles were placed on ice. The pump was then replaced in the well, and the well was left to recover from this disturbance. After ~2 hr, we began purging the well according to the EPA's low-flow purging and sampling protocol as described previously. The unfiltered low-flow samples were collected in 125-mL acid-cleaned HDPE bottles containing diluted nitric acid (as described previously). The samples that were to be filtered in the laboratory were collected in 250-mL acid-cleaned HDPE bottles (with no acid preservative). After the sample bottles were filled, they were placed on ice in a cooler.

However, because it took several hours for the wells to recover from the disturbance caused by removing the pump and Snap Samplers from the wells, all the subsequent samples collected in the first round of sampling were taken using the following sampling order. The Snap Samplers were triggered but left in the well, and the low-flow samples were then collected immediately after the Snap Samplers were triggered. Once the low-flow samples had been collected, the pump was removed from the well, the RGC sampler was removed from the pump, an aliquot of the RGC sample was transferred to a sample bottle, and the sample was acidified and placed on ice as described previously. Finally, the Snap Sampler line was removed from the well, the Snap Sampler bottles were removed from the sampler bodies, the springs were removed from the bottles, the bottles were capped, and the bottom sample bottle was acidified.

For the remaining two wells that were sampled in round two, the following sampling protocol was followed. The Snap Samplers were triggered but left in the well, and the low-flow samples were collected as described previously. The (bundled) pump and Snap Sampler trigger line were removed from the well, the RGC sampler was removed from the pump, an aliquot from the RGC sample was transferred to a sample bottle, and the sample was placed on ice. The Snap Sampler bottles were then removed from the sampler bodies, the springs were removed from the bottles, the bottles were capped, the bottom sample was acidified, and the samples were placed on ice.

All samples were kept on ice until they were delivered to the laboratory the next day.

Sample preparation and chemical analyses

A commercial, EPA-certified laboratory (Eastern Analytical Inc.) was used for the analyses of all the samples collected during this demonstration. Upon arrival at the laboratory, any samples that were to be filtered were filtered with a 0.45 µ Chromafil polyamide (nylon) filter into a 2-ounce HDPE container containing 200 µL of a 1:1 trace-metals grade nitric acid/DI water solution.

For the unfiltered samples, the samples were digested before analyses. This was accomplished by adding 1 mL of trace-metals grade concentrated nitric acid to 50 mL of the test solution and then digesting the sample on a block heater at 85°C until the volume was reduced by one-half. The sample was then cooled to room temperature and brought back to a 50-mL volume by adding reagent-grade water.

Analyses were conducted using inductively coupled plasma/mass spectrometry (ICP/MS) using EPA method 200.8 (Creed et al. 1994) according to EPA SW 846 method 6020B (U.S. EPA 1996).

Statistical analyses

Standard statistical analyses were used to determine if there were statistically significant differences (at the 95% confidence level, $\alpha = 0.05$) between the concentrations of the inorganic analytes in samples collected using the Snap Sampler, the RGC sampler, and low-flow purging and sampling. In all cases, these analyses were conducted on an analyte-by-analyte basis. Analytes with measurable concentrations that allowed statistical analyses included As, Ca, Fe, Mg, Mn, K, and Na. In the few instances where analyte concentrations were below the detection limit, half the detection limit was used in the statistical analyses.

In instances where the data were normally distributed and the variances were homogenous, a repeated measures analysis of variance (RM-ANOVA) test was used. If the data were not normally distributed and/or the variances were not homogeneous, the data were log transformed and retested for normality and homogeneity. If the log-transformed data met the requirements for normality and homogeneity, then the log-transformed data were tested for differences using the RM-ANOVA test. If neither the raw data nor the log-transformed data met the requirements for normality and homogeneity, then a Freidman RM-ANOVA on ranks test was used.

Filtered and unfiltered samples were analyzed separately. Specifically, analyte concentrations in the filtered Snap Sampler samples, the filtered low-flow samples, and the RGC samplers were compared. Concentrations in the unfiltered Snap Sampler samples, the unfiltered low-flow samples, and the RGC samples were also compared.

4 Results and Discussion

Laboratory studies

First study

The purpose of this study was to determine if the Snap Sampler could recover representative concentrations of dissolved metal species and the time needed for equilibration to occur between the sampler materials and the test solution. Initial concentrations of As, Cd, Cr, and Pb in the test solution were ~200 µg/L and concentrations of Ni and Zn were ~400 µg/L. One set of samples (four control samples and one trigger line with three Snap Sampler bottles) was collected 24, 48, and 72 hr after they were deployed in the standpipe (Table 2).

After 24 hr, the Snap Sampler was able to recover more than 99% of the As and Cd and more than 90% of the Cr, Pb, and Ni. There was no statistically significant difference between the concentrations in the control samples and the Snap Sampler samples for any of these analytes. In contrast, recovery of Zn was ~80% with the Snap Sampler, and there was a statistically significant difference between the concentrations of this analyte in the Snap Sampler versus those in the control samples.

After 72-hr equilibration, the percent recoveries were very similar to those observed previously. We were surprised that the percent recovery of Pb, Ni, and Zn had not increased with an increasing equilibration time, so we decided to recover another set of samplers that had remained in the standpipe for 50 days.

Again, there was no statistically significant difference in the analyte concentrations between the control and Snap Sampler samples for the As, Cd, and Cr. However, concentrations of Pb, Ni, and Zn were statistically significantly higher in the control samples than in the Snap Sampler samples. Because we had anticipated that equilibration should have occurred well before 50 days, we examined the concentration data. For the control samples, concentrations of As, Cd, and Cr were similar after 50 days to the initial concentrations in the standpipe. However, this was not the case for Pb, Ni, and Zn. The concentration of Pb was ~15% higher after 50 days than it was initially, the concentration of Ni was almost 30% higher, and the

Table 2. Mean concentrations of metals in Snap Sampler and control samples.

Sample	Mean Concentrations (µg/L)											
	Arsenic		Cadmium		Chromium		Lead		Nickel		Zinc	
	Control	Snap	Control	Snap	Control	Snap	Control	Snap	Control	Snap	Control	Snap
24-hr sample												
Mean concentration (µg/L)	195	195	209	210	196	193	197	188	373	353	438	343
Recovery (%)		100.0		100.7		98.5		95.1		94.5		78.3
% RSD	0.4	0.5	1.3	0.0	0.0	0.5	2.0	0.3	3.9	0.4	5.4	0.5
48-hr sample												
Mean concentration (µg/L)	195	194	210	206	196	193	198	187	387	355	428	346
Recovery (%)		99.6		98.0		98.4		94.3		91.7		80.9
% RSD	0.3	0.8	0.8	0.5	0.8	1.1	1.1	0.6	2.3	0.3	2.7	0.3
72-hr sample												
Mean concentration (µg/L)	194	194	207	208	194	193	199	188	398	365	441	351
Recovery (%)		100.1		100.5		99.5		94.4		91.9		79.6
% RSD	0.3	0.8	0.8	0.5	0.8	1.1	1.1	0.6	2.3	0.3	2.7	0.3
50-day sample												
Mean concentration (µg/L)	188	190	204	205	192	192	231	207	478	447	1031	714
Recovery (%)		100.9		100.7		100.2		89.5		93.4		69.2
% RSD	0.5	0.3	0.5	0.6	0.6	0.6	3.8	0.5	3.8	0.3	7.1	0.3
102-day sample												
Mean concentration (µg/L)	187	186	204	205	198	224	231	225	478	496	1225	1117
Recovery (%)		99.9		100.6		113		97.3		104		91.2
% RSD	0.4	1.2	0.3	0.6	0.0	0.6	2.4	0.7	0.4	0.9	4.0	1.8

concentration of Zn was more than twice the original concentration. Because these analyte concentrations were substantially higher than they were in the initial test solution and because they were also substantially higher than the concentrations in the Snap Sampler samples (taken from the middle of the standpipe), this suggested to us that perhaps the metal spigot on the standpipe had leached these analytes into the control samples.

To verify these results and to determine what further changes there were with time, the final set of samples were collected after 102 days equilibration. This time, the mean concentrations of the Ni and Pb were similar to the values found after 50 days. In contrast, the concentration of Zn was again higher than the concentration found after 50 days (Table 2).

Because we suspected that there was leaching from the sampling spigot, we took an additional set of controls from inside the standpipe on the last sampling day. We did this by lowering a VOA vial on a line and called these samples “bottle controls.” The sampling sequence was as follows: bottle control, spigot control, Snap Sampler, bottle control, and spigot control. Table 3 shows the results from this sampling event. Although these data are somewhat difficult to sort out (no doubt because the standpipe was mixed with each sampling event), it is clear that the concentration of Zn was higher in both spigot controls than it was in either the bottle controls or Snap Sampler. These results support our hypothesis that there was leaching of metals, especially Zn, from the sampling spigot.

Table 3. Concentrations of metals (µg/L) in samples after 102 days of equilibration.

Sample	Arsenic	Cadmium	Chromium	Lead	Nickel	Zinc
Bottle control 1	154	170	165	187	399	941
Spigot control 1	186	204	198	235	479	1260
Snap Sampler	186	205	224	225	496	1117
Bottle control 2	187	203	198	223	478	1140
Spigot control 2	187	203	198	227	476	1190

For all the data, the precision of the data (i.e., agreement between the replicate samples) was excellent (Table 2), with percent relative standard deviations (% RSDs) that were less than 7.5%. However, it is interesting that for As, Cd, and Cr, the % RSDs were always less than 1.5%. In contrast, the % RSD for the metals that we believed had been leached from the sampling spigot (i.e., Pb, Ni, and Zn) were generally greater for the control samples (where the % RSD ranged from 1.1% to 7.1%) than for the Snap Sampler (where the % RSD ranged from 0.3% to 1.8%). These findings also support the hypothesis that there was leaching from the sampling valve into the control samples.

Based upon these findings, we concluded that representative concentrations of As, Cd, and Cr can be obtained with the Snap Sampler. Although there was no definitive evidence that the Snap Sampler can recover representative concentrations of Pb, Ni, and Zn because of the issues with the control samples, we suspect that the results for these analytes would have been similar to those found for As, Cd, and Cr.

Second laboratory study

The purpose of this study was to determine if the Snap Sampler could recover representative concentrations of a broader range of inorganic species, including both cations and anions, and to determine the equilibration time for this process.

Cations that were included in this study included Ca, K, Mg, Mn, and Na. The Snap Sampler was able to recover most (97% to 100%) of the cations after 24 hr and virtually all (more than 99%) of the cations after 48 hr (Table 4). Furthermore, the agreement between duplicate samples was excellent; the percent relative standard deviation was less than 3% (Table 4).

For the anions (i.e., bromide, chloride, nitrate, perchlorate, and sulfate), the Snap Sampler yielded recoveries that were equivalent to those found with the control samples after 24 hr, with the exception of chloride (Table 5). Recovery of the nitrate, perchlorate, and sulfate was 100% and bromide was more than 99%, while recovery of the chloride was 96%. After 48 hr, recovery of all the anions was 100% or better. There was good agreement between duplicate samples with less than 10% RSD (Table 5).

Table 4. Recovery of cations in the second laboratory study.

Sample	Cadmium		Potassium		Magnesium		Manganese		Sodium	
24-hr sample										
Mean concentration (mg/L)	19.4	19.1	42.5	42.7	8.5	8.3	36.1	35.5	47.0	46.6
Recovery (%)		98.7		100.4		97.4		98.3		99.2
% RSD	1.1	0.7	2.0	1.5	0.6	0.5	0.0	0.7	0.5	0.1
48-hr sample										
Mean concentration (mg/L)	19.1	19.2	41.8	42.6	8.3	8.3	35.7	35.5	46.2	46.4
Recovery (%)		100.3		102		99.8		99.4		100.5
% RSD	0.7	0.4	0.5	1.3	0.3	0.1	0.2	0.6	0.8	0.6
72-hr sample										
Mean concentration (mg/L)	19.3	19.1	42.7	42.0	8.35	8.34	35.7	35.7	46.5	46.6
Recovery (%)		99.0		98.5		99.9		100.0		100.0
% RSD	0.7	1.5	1.8	0.0	2.9	2.0	1.6	0.4	0.2	0.8

Table 5. Recovery of anions in the second laboratory study.

Sample	Bromide		Chloride		Nitrate		Perchlorate		Sulfate	
24-hr sample										
Mean concentration(mg/L)	79.5	78.8	66.0	63.5	13.3	13.3	52.9	53.3	61.0	62.0
Recovery (%)		99.1		96.2		100.0		100.8		102
% RSD	0.9	1.3	6.4	5.6	0.5	0.5	8.2	0.9	0.0	0.0
48-hr sample										
Mean concentration (mg/L)	80.0	80.5	64.0	68.5	13.3	13.4	49.7	52.7	61.5	63.0
Recovery (%)		100.6		107		100.8		106		102
% RSD	0.0	0.9	0.0	9.3	0.5	0.5	0.4	3.2	1.1	2.2
72-hr sample										
Mean concentration (mg/L)	81.0	80.0	64.0	64.5	13.4	13.3	47.4	52.5	62.0	62.5
Recovery (%)		98.8		100.8		99.6		111		100.8
% RSD	0.0	0.0	0.0	3.3	0.5	0.0	0.0	0.1	0.0	1.1

Laboratory study conclusions

Currently the ITRC Diffusion/Passive Samplers team (ITRC 2006, 2007) recommends that most passive samplers be left in a well to equilibrate for at least 2 weeks. This allows the well time to recover from the disturbance caused by placing the sampler in the well and gives the formation time to reestablish the normal flow pattern in the well. The findings from these studies demonstrate that this guidance provides more than enough time for the materials in the Snap Sampler to equilibrate with both anions and cations, including metals, in the well water.

5 Field Demonstration at Former Pease AFB

Preliminary field findings

Well redevelopment

When we redeveloped the wells, we were not able to obtain clear water in several wells. Specifically, substantial turbidity remained in the pumped water following redevelopment in well numbers 32-5020 and 32-6135. Also, there was slight turbidity in the water from well 13-6095. Two of these wells, 32-6135 and 13-6095, were bedrock wells. Two of the wells, 13-6095 and 32-5020, were relatively shallow wells. Well 32-6135 had the longest (15 ft) screen.

Flow patterns in the wells

Ambient (nonpumping) flow test data indicated that there was measurable vertical flow in only one of the eight wells, well 32-5031 (Table 6). However, the velocity of the flow in this well was slight, just above the detection capability of this method for this well (0.01 gpm), and was in a negative direction. Thus, under normal ambient conditions (where there is no pumping), there was a slight downward vertical flow in this well. Generally, one expects to find vertical flow in the wells with longer screens (especially those that are 20 ft or more) where there is a difference in hydraulic head. Given that the well screens in this study were all less than 20 ft in length, it is not surprising that we did not have vertical flow in most of these wells. However, it is interesting that the one well where we found slight vertical flow was actually the well with the shortest screen length, 5 ft.

Table 6. Ambient (nonpumping) flow testing results.

Well	Screen*	Probe Position*	Vertical Flow?
32-6064	36.2–46.2	41.3	None detected
32-5020	15.8–26.5	21.1	None detected
32-5031	14.9–19.9	17.4	-0.48 cm/min
32-5076	38.0–48.0	43.0	None detected
32-6135	46.9–61.9	54.5	None detected
32-6008	39.4–53-4	46.4	None detected
13-5045	6.3–14.3	10.3	None detected
13-6095	13.2–23.2	18.4	None detected
*Measurements in feet below top of casing.			

Flow testing during pumping was also conducted, and these results are presented in Table 7. The flow measurements indicated a variety of flow patterns in the test wells. Some wells showed nearly equivalent contributions from the top and bottom portions of the well; e.g., wells 32-5020 and 32-6135. Three of the four bedrock wells showed a significant contribution from one portion of the screened interval under pumped conditions. Specifically, wells 32-6008 and 32-6064 showed significant contributions from the upper zone, and wells 13-6095 and 32-5076 (not a bedrock well) showed significant contributions from the deeper portion of the well. The results for well 32-5031 were anomalous in that the flow rate for the full casing was less than the flow rate for the bottom half of the well. This well was the only well with detectible ambient vertical flow and also had the shortest (5-ft) screen. It is unclear what the mechanism for the discrepancy was, but the ambient vertical flow component likely contributed to the ambiguity in this case. Some of the data for well 13-5045 were lost, which resulted in unreliable whole-screen versus midscreen flow rates.

Table 7. Active pumping flow testing results.

Well	Date	Screen*	Probe Positions*	Flow Rate (mL/min)	Flow %, Bottom One-Half	Flow %, Top One-Half
32-6064	5/21/07	36.2–46.2	41.3 / 34.9	400	10	90
32-5020	5/21/07	15.8–26.5	21.1 / 15.0	400	55	45
32-5031	5/22/07	14.9–19.9	17.4 / 13.9	400	—	—
32-5076	5/22/07	38.0–48.0	43.0 / 37.0	260	70	30
32-6135	5/22/07	46.9–61.9	54.5 / 46.0	170	45	55
32-6008	5/23/07	39.4–53-4	46.4 / 38.4	430	5	95
13-5045	5/23/07	6.3–14.3	10.3 / 5.8	400	—	—
13-6095	5/23/07	13.2–23.2	18.4 / 12.7	400	80	20

*Measurements in feet below top of casing.

Results from the quality assurance and quality control sampling and analyses

The results for all the quality assurance/quality control (QA/QC) samples for all the sampling events, including the preliminary field sampling events are summarized in this section. The results for the trip blanks, laboratory control samples, matrix spikes (MS), and matrix spike duplicates (MSD) are given in Appendix B (see Table B1) and are summarized in Tables 8, 9, and 10.

Table 8. Summary of findings for laboratory control (reference) samples.

	Percent Recovery						
	Arsenic	Cadmium	Iron	Magnesium	Manganese	Potassium	Sodium
	101	109	95	104	90	94	105
	104	109	101	98	91	102	95
	102	100	108	95	100	106	96
	107	103	110	108	109	102	110
	106	103	99	105	99	108	105
	107	103	110	108	109	102	110
	110	104	109	104	104	105	106
Mean	105	104	105	103	100	103	104
Standard deviation	3.147	3.359	6.133	4.914	7.740	4.499	6.094
% RSD	3.0	3.2	5.9	4.8	7.7	4.4	5.9

Table 9. Summary of findings for matrix spike and matrix spike duplicate samples.

Sample		Percent Recovery of Spike						
		Arsenic	Cadmium	Iron	Magnesium	Manganese	Potassium	Sodium
Snap Sampler	MS	104	108	97	108	82	101	106
	MSD	106	103	102	104	84	100	107
Low-flow	MS	110	111	96	99	95	91	117
	MSD	100	101	93	96	88	90	111
Low-flow	MS	111	99	103	95	93	91	95
	MSD	111	93	98	90	87	93	99
Low-flow	MS	121	127	99	110	97	107	111
	MSD	115	123	100	104	94	106	109
Low-flow	MS	115	126	94	119	101	122	123
	MSD	113	130	94	119	101	115	113
Low-flow	MS	119	85	105	120	101	118	124
	MSD	119	96	103	121	103	116	123
Low-flow	MS	110	96	98	92	102	96	95
	MSD	108	96	97	95	104	96	96
Mean		112	107	99	105	95	103	109
Standard deviation		5.9835	14.467	3.7159	11.162	7.4095	11.066	10.297
% RSD		5.4	13.6	3.8	10.6	7.8	10.7	9.4

Table 10. Summary of findings for laboratory and field duplicates.

		Percent Relative Standard Deviation						
		Arsenic	Cadmium	Iron	Magnesium	Manganese	Potassium	Sodium
Laboratory duplicates	Mean	2.46	2.71	1.27	4.52	5.32	2.76	3.36
	Range	0–4.7	0–4.9	0–4.0	1.8–8.7	0–14.8	0–4.6	0–10.3
Field duplicates	Mean	2.93	4.24	4.95	1.57	3.33	3.59	4.56
	Range	0–3.9	0–10.5	0–10.9	0–9.4	0–5.5	0–6.1	0–9.4

Generally, the QA/QC data was excellent. For the trip blank data, concentrations were below the detection limit for all of the analytes (see Table B1). For the laboratory control samples, the percent recovery was within 10% of the known (reference) value in all cases (Table 8). For the MS and MSD samples, recovery was within 20% of the spiked value for most of the analytes (Table 9). Ca, Mg, K, and Na were exceptions where the percent recovery was as high as 130% for Ca, 121% for Mg, 122% for K, and 124% for Na.

The data for the laboratory duplicates and the field duplicates are given in Appendix B (see Tables B2 and B3). These data are summarized in Table 10. Generally, the agreement between the laboratory duplicates and the field duplicates was excellent. The percent relative standard deviation was 5% (or less) in most cases and 10% (or less) in all but one instance.

The data for the equipment blanks can be found in Appendix B (see Table B4). The equipment was tested to determine whether any of the materials in the samplers leached any of the analytes at concentrations that could interfere with the analyses. Equipment that was tested included all the bladder pumps used in the study and the RGC sampler and Snap Sampler samples. Concentrations of all the analytes were below the detection limit in all but a few instances and, in those cases, the concentrations were all below 0.3 mg/L, which was well below concentrations found in the samples.

Depth profiles of analyte concentrations in wells

Table 11 shows the results from vertically profiling analyte concentrations in the wells. For most of the wells, there was no substantial difference between the analyte concentrations in the samples collected from the shallow versus the deeper portion of the well. The most notable exception was well 32-5020, where concentrations of all the analytes, except Na, were higher in the deeper part of the well. However, we believe that this difference was caused by the heavier sediment load found in the deeper sample (Figure 7). This is one of the wells we were not able to successfully redevelop. We suspect that the filter pack and/or screen for this well may not have been correctly sized or had changed with time due to corrosion of the stainless steel screen in this well and thus was unable to prevent fines from entering the well.

Two of the bedrock wells also showed concentration gradients with depth for As and Fe. Wells 32-6064 and 13-6095 both had slightly higher concentrations of these analytes in the shallow portion of the well.

Comparative field demonstration

The purpose of this field demonstration was to determine if the Snap Sampler recovered samples with equivalent concentrations of inorganic analytes to those recovered in samples collected using low-flow purging and sampling. In addition, we also wanted to examine the relationship between analyte concentrations of inorganics in samples collected using the Snap Sampler, low-flow sampling, and a diffusion sampler (i.e., the RGC sampler). We anticipated that the concentrations of these analytes in whole-water samples (collected either with the Snap Sampler or low-flow sampling) would most likely be higher than concentrations in a diffusion sampler (which would be naturally filtered by the membrane) if colloidal transport was significant in these wells.

The metals found in measurable concentrations in these wells included Ca, Fe, Mg, Mn, K, and Na. These metals typically occur as cations in ground water. In addition, these wells contained measurable concentrations of As, which is usually present in anionic complexes in ground water.

Given that there were not large differences in analyte concentrations with depth in these wells, we would predict that we would not expect to find large differences between analyte concentrations in the Snap Sampler samples versus those collected using low-flow purging and sampling.

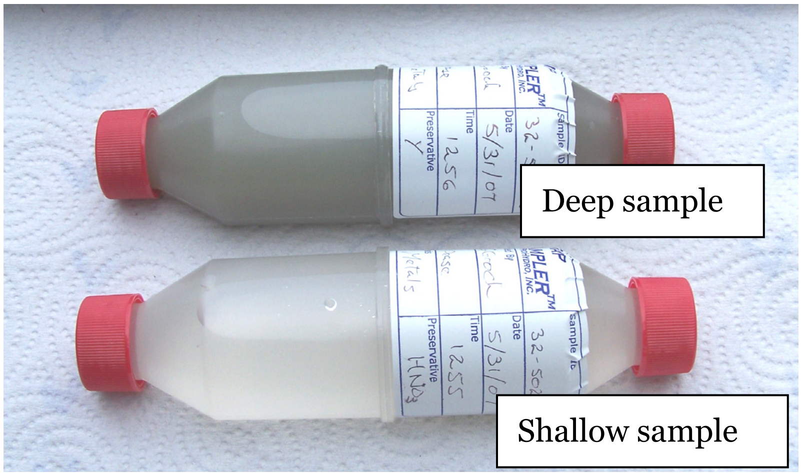


Figure 7. Samples collected from deep and shallow portions of well 32-5020.

Equipment performance

Generally, the Snap Samplers performed well by triggering on demand and providing full samples. Out of the 36 samples collected, there were three instances when one of the caps did not seal properly because the o-ring did not seat properly. In these instances, there was sample leakage from the sampler as it was withdrawn from the well. There were two bottles that had pinhole leaks in the seam. (These bottles were returned to the developer after the sampling was complete so that they could be examined. The developer has since changed the polymer used in the bottles from PP to HDPE. This polymer has better weld properties, is less likely to become brittle at low temperatures such as those used to ship the samples, and should be more resistant to cracking.) There also was one instance where we found the top cap of one of the samplers deployed prematurely, and there was one instance during deployment where the ball connector had pulled off the trigger line and the cable had retracted back into the outer tubing. Because there were issues with ~10% of the samples collected, it might be prudent to add an additional bottle to the trigger line as added insurance.

Table 11. Results from the depth profiles of the wells.

Well	Depth	Concentration (mg/L)						
		Arsenic	Cadmium	Iron	Magnesium	Manganese	Potassium	Sodium
13-5045	Shallow	0.15	67	17	11	5.4	2.5	80
	Deep	0.14	61	17	12	6.0	2.9	110
13-6095	Shallow	0.140	29	7.9	6.3	3.9	1.2	10
	Deep	0.065	28	3.7	5.9	3.7	1.1	11
32-5020	Shallow	0.15	190	9.8	44	0.53	8.4	60
	Deep	0.25	230	160	82	2.9	33	60
32-5031	Single depth	0.11	89	12	21	0.40	5.2	30
32-5076	Shallow	0.093	65	7.4	14	0.39	4.6	30
	Deep	0.089	52	8.8	14	0.36	4.9	30
32-6008	Shallow	0.066	65	1.5	24	0.15	7.3	70
	Deep	0.057	69	1.2	24	0.15	7.2	70
	Deep duplicate	0.060	65	1.3	24	0.15	7.2	80
32-6064	Shallow	0.11	150	2.6	42	0.27	9.2	50
	Deep	0.03	140	0.2	40	0.34	9.0	60
32-6135	Shallow	0.022	3.7	2.1	1.6	0.038	6.2	120
	Deep	0.021	4.5	2.9	1.6	0.059	6.4	130

In contrast, we had to purge some of the wells for more than an hour. Part of the problem was that the flow-through cell collected particulates. To correct this problem, we had to keep cleaning the flow-through cell and each time we cleaned it, we had to hook the equipment back up and then wait for the purge parameters (such as temperature and DO) to restabilize. In some cases, this increased the purge time. Furthermore, on one of our sampling days, we had problems with the meter readings. After purging the well to what we thought was a clear sample, the turbidity read 62 (using the Horiba meter). We then collected a sample and used the LaMotte turbidity meter, where the reading was 6.5. Our recommendation for future sampling events would be not to obtain turbidity readings using a flow-through cell, but rather to collect the samples separately and measure them with a meter such as the Horiba meter.

Unfiltered samples

The findings for this study can be found in Table 12. There was not a statistically significant difference in the concentrations of these analytes in the diffusion sampler versus those in the samples collected using low-flow purging and sampling or the Snap Sampler, with one exception—the Fe levels in the Snap Sampler. Given that 1) the small pore size (18 Å or 1.8 x 10^{-3} µ) of the cellulose membrane prevented all but the smallest of colloidal particles from entering the RGC sampler and 2) the low-flow and Snap Sampler samples were whole-water samples, these findings indicate that colloidal transport of these contaminants is not a predominant transport mechanism at this site.

Table 12. Findings from the field study.

		Filtered Samples			Unfiltered Samples		
		Mean Concentration (mg/L)			Mean Concentration (mg/L)		
Analyte	Well	Low-flow	RGC	Snap	Low-flow	RGC	Snap
Arsenic	13-5045	0.097	0.16	0.12	0.16	0.16	0.16
	13-6095	0.044	0.059	0.054	0.056	0.059	0.058
	32-5020	0.074	0.15	0.053	0.15	0.15	0.15
	32-5020	0.068	0.094	0.045	0.11	0.094	0.10
	32-5031	0.031	0.11	0.024	0.063	0.11	0.13
	32-5076	0.026	0.062	0.019	0.054	0.062	0.051
	32-6008	0.051	0.055	0.041	0.059	0.055	0.069
	32-6064	0.060	0.092	0.022	0.095	0.092	0.14
	32-6064	0.070	0.10	0.043	0.093	0.10	0.11
	32-6135	0.025	0.023	0.027	0.023	0.023	0.027
	Mean	**0.055**[a]	**0.091**[b]	**0.045**[a]	**0.086**[c]	**0.091**[c]	**0.100**[c]

Table 12 (cont'd). Findings from the field study.

		Filtered Samples			Unfiltered Samples		
		Mean Concentration (mg/L)			Mean Concentration (mg/L)		
Analyte	Well	Low-flow	RGC	Snap	Low-flow	RGC	Snap
Cadmium	13-5045	69	71	70	72	71	66
	13-6095	43	43	42	42	43	41
	32-5020	230	250	230	230	250	190
	32-5020	140	130	150	150	130	150
	32-5031	72	86	93	75	86	97
	32-5076	62	58	60	58	58	53
	32-6008	97	98	96	98	98	100
	32-6064	170	170	170	170	170	180
	32-6064	110	110	110	110	110	110
	32-6135	4.5	4.4	4.6	4.3	4.4	5.1
	Mean	100^{a}	102^{a}	103^{a}	101^{c}	102^{c}	99^{c}
Iron	13-5045	9.1	14.0	9.4	15	14	14
	13-6095	1.5	3.1	2.4	3.3	3.1	3.7
	32-5020	<0.05	4.4	<0.05	5.0	4.4	13
	32-5020	<0.05	2.6	0.14	3.6	2.6	4.1
	32-5031	<0.05	9.0	<0.05	4.4	9.0	20
	32-5076	<0.05	4.4	<0.05	2.3	4.4	6.3
	32-6008	<0.05	0.8	<0.05	0.6	0.8	1.4
	32-6064	<0.05	1.8	<0.05	1.8	1.8	4.5
	32-6064	<0.05	2.1	<0.05	1.8	2.1	2.6
	32-6135	<0.05	0.25	<0.05	0.3	0.3	4.5
	Mean	1.1^{a}	4.2^{b}	1.2^{a}	3.8^{c}	4.2^{c}	7.4^{d}
Magnesium	13-5045	12.0	12.0	13.0	12.0	12.0	12.0
	13-6095	8.1	7.7	8.0	7.6	7.7	7.5
	32-5020	53	52	53	53	52	47
	32-5020	38	36	40	36	36	41
	32-5031	18	21	22	18	21	25
	32-5076	13	13	13	13	13	12
	32-6008	35	34	33	34	34	33
	32-6064	49	52	51	52	52	50
	32-6064	43	40	38	38	40	36
	32-6135	1.9	1.8	1.9	1.8	1.8	2.2
	Mean	27^{a}	27^{a}	27^{a}	27^{c}	27^{c}	27^{c}

Table 12 (cont'd). Findings from the field study.

		Filtered Samples			Unfiltered Samples		
		Mean Concentration (mg/L)			Mean Concentration (mg/L)		
Analyte	Well	Low-flow	RGC	Snap	Low-flow	RGC	Snap
Manganese	13-5045	8.0	7.8	7.8	7.3	7.8	7.4
	13-6095	8.0	7.9	7.8	7.9	7.9	7.7
	32-5020	0.70	0.71	0.70	0.71	0.71	0.77
	32-5020	0.43	0.44	0.49	0.46	0.44	0.54
	32-5031	0.23	0.46	0.44	0.23	0.46	0.75
	32-5076	0.53	0.56	0.53	0.52	0.56	0.55
	32-6008	0.22	0.22	0.21	0.22	0.22	0.22
	32-6064	0.37	0.41	0.38	0.37	0.41	0.44
	32-6064	0.20	0.21	0.21	0.20	0.21	0.21
	32-6135	0.018	0.024	0.022	0.024	0.024	0.130
	Mean	**1.9**[a]	**1.9**[a]	**1.9**[a]	**1.8**[c]	**1.9**[c]	**1.9**[c]
Potassium	13-5045	2.8	2.7	2.7	2.8	2.7	2.6
	13-6095	1.6	1.6	1.6	1.5	1.6	1.5
	32-5020	8.6	8.4	8.6	8.8	8.4	8.7
	32-5020	7.1	6.4	7.0	7.1	6.4	7.6
	32-5031	5.3	4.6	4.8	5.4	4.6	6.8
	32-5076	4.0	3.8	3.9	3.8	3.8	4.0
	32-6008	10.0	9.8	10.0	10.0	9.8	10.0
	32-6064	11.0	12.0	11.0	11.0	12.0	12.0
	32-6064	9.8	9.7	9.2	9.0	9.7	8.9
	32-6135	8.20	8.40	8.30	7.9	8.4	8.2
	Mean	**6.8**[a]	**6.7**[a]	**6.7**[a]	**6.7**[c]	**6.7**[c]	**7.0**[c]
Sodium	13-5045	120	98	99	140	98	88
	13-6095	15	14	14	14	14	14
	32-5020	73	72	72	72	72	62
	32-5020	57	54	62	56	54	62
	32-5031	59	34	34	60	34	32
	32-5076	27	26	27	26	26	23
	32-6008	99	92	94	95	92	91
	32-6064	68	69	70	71	69	68
	32-6064	61	58	55	53	58	55
	32-6135	160	160	160	180	160	170
	Mean	**74**[a]	**68**[b]	**69**[a,b]	**77**[c]	**68**[c]	**67**[c]

[a,b,c] For each analyte, there was no statistically significant difference between values with the same letter.

We believe that the Fe concentrations in the Snap Sampler represented the conditions in the well at the time of sampling and are an indication of the higher levels of particulates that were present in the well at the time of sampling as seen by examining the initial turbidity values during low-flow sampling (Table 13). Generally, the initial turbidity values were considerably lower in most of the wells during the preliminary round of low-flow sampling than they were during the field demonstration. We believe that placing all the additional equipment in the well (i.e., the baffle, pump, Snap Samplers, RGC samplers, and bottle weight) acted as a surge block in the well and elevated turbidity and oxygen levels in the well water (which caused oxidation/precipitation reactions to occur with the iron present in the well water), and thereby falsely elevated concentrations of Fe. Apparently, some of the wells never fully recovered, even after leaving this equipment in the wells for two weeks. The most affected wells included 32-5020, 32-5031, and 32-5076. We will discuss this in more detail when we analyze this data on a well-by-well basis.

Filtered samples

For the filtered samples, there was no statistically significant difference between the analyte concentrations in the Snap Sampler samples and the low-flow samples for any of the analytes. There also was not a statistically significant difference between the concentrations of Ca, Mg, Mn, and K in the RGC samples when compared with the filtered low-flow samples. However, there was a statistically significant difference between the concentrations of As, Fe, and Na in the RGC samples and the filtered low-flow samples; concentrations of As and Fe were higher in the RGC samples and concentrations of Na were lower. These anomalies can best be understood by comparing the concentrations in the filtered and unfiltered low-flow samples (discussed below).

Filtered versus unfiltered samples

When the low-flow filtered samples were compared with the low-flow unfiltered samples, there was a statistically significant difference between the treatments for As and Fe, but not for the other cations. If colloidal transport is not responsible for these differences (as shown previously) then the filtered samples must have a falsely low bias. Since these samples were filtered in the laboratory, it is reasonable to assume that there were losses of iron due to precipitation of hydrous iron oxides and that the losses of the As anion resulted from coprecipitation with the hydrous iron oxides. Sev-

Table 13. Purge times and turbidity values[1] for low-flow sampling for the initial sampling and the field demonstration.

	Well No.	32-6064	32-6064	32-5020	32-5020	32-5031	32-5076	32-6135	13-5045	32-6008	13-6095
	Pump No.	2291	2291	2269	2269	1470	2271	2270	2292	1472	1400
Initial Sampling	Purge time (min)	88		37		61	79	91	61	63	50
	Initial turbidity	2		4		14	11	27	8	31	9
	Final turbidity	1		4		11	8	20	8	32	11
Field Demonstration	Round	1	2	1	2						
	Purge time (min)	99†	86	62†	223	121	214	81	81	63	93
	Initial turbidity	15†	7	40†	11	490	38	22	11	9	14
	Final turbidity	5†	4	42*†	26*	11	20	6.5**	9	9	7

[1] All turbidity values taken with Horiba probe unless noted otherwise.

†Purge times and turbidity readings are greater because the sampling equipment was removed from the well and then the pump was replaced in the well before collecting the low-flow samples.

*Issues with meter and with particles collecting in flow-through cell.

**Value measured with LaMotte turbidity meter.

eral studies have shown that anions are more readily removed by iron oxides than cations (Jenne 1968; Korte et al. 1976; Korte and Fernando 1991) and that the most important mechanism for the accumulation of arsenic in sediment is adsorption on and coprecipitation with hydrous iron oxides (Aggett and O'Brien 1985 and Aggett and Roberts 1986 [as given by Mok and Wai 1990]; Korte 1991). The findings between the filtered and unfiltered Snap Sampler samples were also similar.

Initially, we had considered using in-line filtration for the low-flow samples. However, the Snap Sampler samples could not be filtered using an in-line filter, and we wanted to use the same filtration equipment for both sampling methods. We considered filtering the samples in the field using a suction or hand filtration apparatus but were concerned that this might add more bias than simply shipping the samples to the laboratory for filtration. Given our findings, we believe that field filtration of the samples would have been prudent. Although the sample is also exposed to air while the vial is filled and during filtration, the contact time for this reaction would be less and presumably any low bias in these samples would be less.

Well-by-well comparisons using the unfiltered sample data

Although the statistical analyses clearly indicated that there was no statistically significant bias associated with using the Snap Sampler when compared with low-flow purging and sampling for almost all of the analytes, we thought we would examine the unfiltered data on a well-by-well basis to see if there were any wells where there appeared to be a large difference between the sampling methods.

The flow testing showed that for four of the wells (32-6008, 32-6064, 13-6095, and 32-5076) there was a significant contribution from one portion of the well under pumped conditions, but there was no detectable vertical flow under ambient conditions. We expected that we might find poor agreement between the concentrations in the Snap Sampler versus the pumped samples for those wells where there also was a concentration gradient with depth in well, i.e., in wells 13-6095 and 32-6064 (where there were higher concentrations of As and Fe in the shallow portion of the well screen). However, there was excellent agreement between the analyte concentrations in the pumped and the Snap Sampler samples in these wells.

In contrast, the wells with the poorest agreement between the concentrations in the unfiltered samples taken with the Snap Sampler versus the low-flow samples were well numbers 32-6135 and 32-5031. For well 32-6135, the concentration of iron was ~10 times higher in the Snap Sampler, and the Mn concentration was ~5 times higher. Well 32-5031 also had higher con-centrations of three analytes (As, Fe, and Mn) in the Snap Sampler sample; As was 2 times higher, Fe was 5 times higher, and the Mn concentration was ~3 times higher. Both of these wells are stainless steel wells. (The other six wells were made with PVC casing and screen.) Although we might expect Fe and Mn concentrations to be higher in stainless steel wells as a result of leaching (Parker et al. 1990; Hewitt 1989, 1992, 1994), arsenic is not a component of stainless steel so there is no reason to expect As concentrations to be higher in the stainless steel wells. Thus, we suspect that much of the elevated concentrations of these analytes resulted from higher turbidities in these wells. Higher turbidities in the wells may be the result of installation of all the sampling equipment in the well (as mentioned previously), from poor well-construction methods, or from degradation of (corrosion of) the well screens resulting in an increased slot size. (The improper selection of the slot size of the screen and/or the grain size of the filter pack can result in larger fines entering a well.) We had not been able to successfully redevelop well 32-6135; this may be an example of a well with an elevated native turbidity that resulted from poor well-construction design or degradation of the screen. In contrast, well 32-5031 appears to have been dramatically affected by the installation of all the equipment in the well; the initial low-flow turbidity value in the well was 35 times higher in this sampling event than it was in the initial sampling round (Table 13).

Other wells where there were elevated iron levels included numbers 32-5076, 32-5020, and 32-6064. These wells also appear to have been affected by the installation of the sampling equipment in the well as the initial turbidity reading for these sampling events were 2 to 10 times higher than during the preliminary sampling round (Table 13).

Generally, concentrations of Fe in the RGC samplers were similar to those found in the low-flow samples, except for the samples from wells 32-5031 and 32-5076 where the concentrations of Fe were elevated but not as much as that found in the Snap Sampler. Concentrations of As and Mn were also similarly elevated in well 32-5031. These data also support the hypothesis that the Snap Sampler concentrations were elevated because of higher turbidities in these wells.

In contrast to the other analytes where occasionally higher concentrations were found in the Snap Sampler, lower concentrations of unfiltered Na were found in the Snap Sampler samples collected from wells 32-5031 and 13-5045, where concentrations were nearly one-half those found in the low-flow samples. This may be because low-flow pumping brought in a different water quality than what was present in the well. Interestingly, these were the two shallowest wells and would be most affected by rainfall events.

6 Conclusions

Laboratory studies demonstrated that the Snap Sampler recovered equivalent concentrations of both anions and cations, including several metals and perchlorate, when compared with known control samples.

In our field study, the Snap Sampler generally recovered samples with equivalent concentrations of inorganic analytes to those found using the EPA's low-flow purging and sampling protocol. This was true for both filtered and unfiltered samples, with the exception of the unfiltered Fe samples. We believe that this exception is the result of the elevated turbidity in some wells. In some cases, we believe that the elevated turbidity was the result of well-construction practices or possibly degradation of the stainless steel screen. However, for most of the wells, the elevated concentrations appear to be the direct result of disturbance of the well resulting from installing the sampling equipment. This is an issue we will examine in more depth at our next test site.

The RGC sampler also provided samples with equivalent concentrations of these inorganic analytes when compared with unfiltered low-flow samples. The use of the RGC sampler in conjunction with low-flow sampling and the Snap Sampler demonstrated that colloidal transport of these analytes was not predominant at this site.

We believe that both passive sampling methods used in this study can be used successfully to sample for most inorganic analytes. However, the Snap Sampler can be left in place between the sampling events and can be used to collect unfiltered and filtered samples. In contrast, the RGC sampler can only be used to collect filtered samples and cannot be left in the well between sampling events.

References

Aggett, J., and G. A. O'Brien. 1985. Detailed model for the mobility of arsenic in lacustrine sediments based on measurements in Lake Ohakuri. *Environmental Science and Technology* 19:231.

Aggett, J., and L. S. Roberts. 1986. Insight into the mechanism of accumulation of arsenate and phosphate in Hydro Lake sediments by measuring the rate of dissolution with ethylenediaminetetraacetic acid. *Environmental Science and Technology* 20:183.

Bailey, R., J. Marchesani, A. C. Marinucci, J. Reynard, and P. Sanders. 2005. Use of sequential filtration for determining transportable lead in ground water. *Ground Water Monitoring and Remediation* 25(3):52–62.

Britt, S. L. 2005. Testing the in-well horizontal laminar flow assumption with a sand tank well model. *Ground Water Monitoring and Remediation* 25(3):73–81.

Britt, S. L. 2006. Multilevel sampling in traditional monitoring wells: a mixing-limited passive sampling approach. In *Proceedings of the Ground Water Resources Association of California High Resolution Site Characterization and Monitoring Conference*, Long Beach, CA, November 14–15, 2006.

Creed, J. T., C. A. Brockhoff, and T. D. Martin. 1994. *Method 200.8: Determination of trace elements in waters and wastes by inductively coupled plasma—mass spectrometry*. Cincinnati, OH: Environmental Monitoring Systems Laboratory, Office of Research and Development, U.S. Environmental Protection Agency. (Rev. 5.4) web1.er.usgs.gov/nemi/method_pdf/4665.pdf

Elci, A., F. J. Molz III, and W. R. Waldrop. 2001. Implications of observed and simulated ambient flow in monitoring wells. *Ground Water* 39(6):853–862.

Gillham, R. W., M. J. L. Robin, F. F. Barker, and J. A. Cherry. 1985. *Field evaluation of well purging procedures*. API Publication 4405. Washington, DC: Environmental Affairs Department, American Petroleum Institute.

Gretener, P. E. 1967. On the thermal instability of large diameter wells—an observational report. *Geophysics* 32(4):727–738.

Hautman, D. P., D. J. Munch, A. D. Eaton, and A. W. Haghani. 1999. *Method 314.0: Determination of perchlorate in drinking water using ion chromatography*. Cincinnati, OH: National Exposure Research Laboratory, Office of Research and Development, U.S. Environmental Protection Agency.

Hewitt, A. D. 1989. *Leaching of metal pollutants from four well casings used for ground-water monitoring*. CRREL Special Report 89-32. Hanover, NH: U.S. Army Cold Regions Research and Engineering Laboratory.

Hewitt, A. D. 1992. Potential of common well materials to influence aqueous metal concentrations. *Ground Water Monitoring and Remediation* 12(2):131–136.

Hewitt, A. D. 1994. Dynamic study of common well screen materials. *Ground Water Monitoring and Remediation* 14(1):87–94.

ITRC. 2004. *Technical and regulatory guidance for using polyethylene bag samplers to monitor volatile organic compounds in groundwater*. Diffusion/Passive Sampler Team DSP-3. Washington, DC: Interstate Technology Regulatory Council. http://www.itrcweb.org/Documents/DSP-3.pdf

ITRC. 2006. *Technology overview of passive sampler technologies*. Diffusion/Passive Sampler Team DSP-4. Interstate Technology Regulatory Council. http://www.itrcweb.org/Documents/DSP-4.pdf

ITRC. 2007. *Protocol for use of five passive samplers to sample a variety of contaminants in groundwater*. Diffusion/Passive Sampler Team DSP-5. Interstate Technology Regulatory Council. http://www.itrcweb.org/Documents/DSP-5.pdf

Jenne, E. A. 1968. Controls on Mn, Fe, Co, Ni, Cu, and Zn concentrations in soils and water: the significant role of hydrous Mn and Fe oxides. *Advances in Chemistry Series* 73:337–387.

Korte, N. 1991. Naturally occurring arsenic in groundwaters of the Midwestern United States. *Environmental Geology* 18(2):137–141.

Korte, N., J. Skopp, W. H. Fuller, E. E. Niebla, and B. A. Alesii. 1976. Trace element movement in soils: influence of soil physical and chemical properties. *Soil Science* 122:350–359.

Korte, N. E., and Q. F. Fernando. 1991. A review of arsenic(III) in groundwater. *Critical Reviews in Environmental Control* 21:1–39.

Michalski, M. 1989. Application of temperature and electrical conductivity logging in ground water monitoring. *Ground Water Monitoring Review* 9(3):112–118.

Mok, W. M., and C. M. Wai. 1990. Distribution and mobilization of arsenic and antimony species in the Coeur D'Alene River, Idaho. *Environmental Science and Technology* 24:102.

MWH Americas Inc. 2004. *Former Pease Air Force Base, Portsmouth, NH: 5-Year Review Report (1999 to 2004)*. Contract No. F41624-03-D-860B, Task Order No. 058, Sept 2004. http://www.epa.gov/superfund/sites/fiveyear/f04-01013.pdf)

New Jersey Department of Environmental Protection. 2005. *Field sampling procedures manual*. Trenton, NJ: New Jersey Department of Environmental Protection.

Parker, L. V., A. D. Hewitt, and T. F. Jenkins. 1990. Influence of casing materials on trace-level chemicals in well water. *Ground Water Monitoring Review* 10(2):146–156.

Parker, L. V., and N. D. Mulherin. 2007. *Evaluation of the Snap Sampler for sampling ground water monitoring wells for VOCs and explosives*. ERDC/CRREL TR-07-14. Hanover, NH: U.S. Army Cold Regions Research and Engineering Laboratory.

Parsons Inc. 2003. *Final comprehensive results report of the passive diffusion bag demonstration*. Prepared for the Air Force Center for Environmental Excellence, Science and Engineering Division, Contract No. F41624-00-D-8024, August 2003.

Parsons Inc. 2005. *Results report for the demonstration of no-purge groundwater sampling devices at former McClellan Air Force Base, California*. Prepared for the U.S. Army Corps of Engineers, Omaha District, Air Force Center for Environmental Excellence, Air Force Real Property Agency, Contract No. F44650-99-D-005, March 2005. http://www.snapsampler.com/images/McClellan_Report_Highlighted.pdf

Pfaff, J. D. 1993. *Method 300.0: Determination of inorganic anions by ion chromatography*. Washington, DC: Environmental Monitoring Systems Laboratory, Office of Research and Development, U.S. Environmental Protection Agency. (Rev. 2.1, August 1993). http://www.epa.gov/waterscience/methods/method/files/300_0.pdf

Powell, R. M., and R. W. Puls. 1993. Passive sampling of ground water monitoring wells with out purging: multilevel well chemistry and tracer disappearance. *Journal of Contaminant Hydrology* 12:51–77.

Puls, R. W., and M. J. Barcelona. 1996. *Low-flow (minimal drawdown) ground-water sampling procedures*. U.S. EPA publication EPA/540/s-95/504. Washington, DC: U.S. Environmental Protection Agency.

Puls, R. W., and C. J. Paul. 1997. Multi-level sampling in conventional monitoring wells for improved estimation of vertical contaminant distributions and mass. *Journal of Contaminant Hydrology* 25:85–111.

Robin, M. J. L., and R. W. Gillham. 1987. Field evaluation on well purging procedures, *Ground Water Monitoring Review* 7(4):85–93.

Ronen, D., M. Magaritz, and I. Levy. 1987. An in-situ multilevel sampler for preventive monitoring and study of hydrochemical profiles in aquifers. *Groundwater Monitoring and Remediation* 7(3):69–74.

Sammel, E. 1968. Convective flow and its effect on temperature logging in small-diameter wells. *Geophysics* 33(6):1004–1012.

U.S. EPA Region I. 1996. *Low stress (low flow) purging and sampling procedure for the collection of ground water samples from monitoring wells, revision 2*. U.S. EPA Region I SOP No. GW 0001. Boston, MA: U.S. Environmental Protection Agency Region I.

U.S. EPA. 1996. *SW-846. Test methods for evaluating solid waste, physical/chemical methods*, 3rd Ed. Washington, DC: Office of Solid Waste, U.S. Environmental Protection Agency. http://www.epa.gov/epaoswer/hazwaste/test/main.htm.

Vroblesky, D. A. 2001. *User's guide for polyethylene-based passive diffusion bag samplers to obtain volatile organic compound concentrations in wells, parts 1 and 2*. USGS Water Resources Investigation Reports 01-4060 and 01-4061. Columbia, SC: U.S. Geological Survey.

Vroblesky, D. A., C. C. Casey, and M. A. Lowery. 2006. *Influence of in-well convection on well sampling*. USGS Scientific Investigation Report 2006-5247. Reston, VA: U.S. Geological Survey.

Vroblesky, D. A., M. Joshi, J. Morrell, and J. E. Peterson. 2003. *Evaluation of passive diffusion bag samplers, dialysis samplers, and nylon-screen samplers in selected wells at Andersen Air Force Base, Guam, March–April 2002*. USGS Water Resources Investigations Report 03-4157. Columbia, SC: U.S. Geological Survey.

Vroblesky, D. A., and B. C. Peters. 2000. *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. Columbia, SC: U.S. Geological Survey.

Vroblesky, D. A., and M. D. Petkewich. 2000. *Field testing of passive diffusion bag samplers for volatile organic compound concentrations in ground water, Naval Industrial Reserve ordnance plant, Fridley, Minnesota, November 1999 and May 2000*. USGS Water Resources Investigations Report 00-4246. Columbia, SC: U.S. Geological Survey.

Appendix A: Site Description of the Former Pease AFB

The following information has been taken from a site report by MWH Americas Inc. (2004).

At the onset of World War II, an airport at the former Pease AFB site was used by the U.S. Navy. The U.S. Air Force assumed control of the site in 1951, and construction of the base was completed in 1956. Over time, various quantities of fuels, oils, lubricants, solvents, and protective coatings were used; as a result of these activities, contaminants from these substances were released into the environment. Specifically, fuels, organic solvents, PAHs, and metals have been found in soils on the base. The ground water has been found to be contaminated with volatile organic compounds (VOCs) including trichloroethylene (TCE) and tetrachloroethylene (PCE). PAHs, pesticides, and heavy metals have been found in the sediments from various discharge ditches.

In 1976, the DoD initiated an assessment of the environmental contamination resulting from the past operation and disposal practices at all DoD facilities. In 1980, in response to the Resource Conservation and Recovery Act (RCRA), and in anticipation of the Comprehensive Environmental Response, Compensation, and Liability Act (CERCLA), DoD issued a memorandum requiring identification of all hazardous waste disposal sites on DoD facilities. In 1984, a Phase I Problem Identification Search was completed at the former Pease AFB that assessed whether potential hazardous waste sites warranted further inspection. As a result of that report, the Air Force was designated as the lead agency to conduct CERCLA cleanup activities.

In December 1988, Pease AFB was selected as one of 86 military installations to be closed by the Secretary of Defense's Commission on Base Realignment and Closure (BRAC). The base was closed as an active installation in March 1991. The Air Force has transferred most of the former AFB to the Pease Development Authority via quitclaim deed. The airfield is now a fully operational commercial airport. Other property is currently being used or developed for light commercial and industrial facilities. A portion of the base was transferred to the U.S. Department of the Interior for use as a national wildlife refuge, and the Air Force retained 229 acres for use by the New Hampshire Air National Guard.

The former base was listed on the National Priorities List (NPL) in 1990. In 1991, the Air Force, EPA, and New Hampshire Department of Environmental Services (NHDES) signed a Federal Facilities Agreement (FFA) establishing the protocols for conducting the environmental study and cleanup of the former AFB. This agreement established eight Installation Restoration Program (IRP) zones for which separate remedial investigation (RI) and feasibility study (FS) reports were prepared.

RI and FS reports were completed in 1994 and were used to develop Record of Decisions (RODs) for the individual IRP zones. Source area RODs were also developed for several sites where interim remedial measures had been implemented. These sites were prioritized by the Air Force as posing significant risk to human health and the environment. The RODs have become the controlling documents for site cleanup at the former Pease AFB. The eight monitoring wells selected for this demonstration were located in Zones 1 and 3. Descriptions of the activity that occurred in these zones and the geology of these zones was taken from the five-year review published in 2004 (MWH Americas Inc. 2004) as follows.

Zone 1

Zone 1 is located in the northeastern portion of the former Pease AFB and contains several IRP sites. These include several landfills (nos. 2, 3, 4, and 5), a paint can disposal area, a bulk fuel storage area (BFSA) (Site 13), and the railroad tracks. Two of the wells selected for this demonstration are located in the BFSA. The BFSA was the main storage area for fuels at the base and has been in operation from 1953 to the present. The BFSA encompasses approximately 16 acres and is located at the northeastern section of this zone.

Major spills at the BFSA occurred in 1963, 1975, and 1980. In March 1993, the site was removed from the FFA because the sources of contamination at the site were petroleum based (which are non-CERCLA regulated wastes). Upon removal from CERCLA, the site was put under the state's jurisdiction.

The overburden deposits across Zone 1 include younger sediments, such as marsh deposits, and older deposits, such as glacial-marine deposits. The unconsolidated stratigraphic units identified are fill, upper sand (US), marine clay and silt (MCS), lower sand (LS), and glacial till. One or more of these units may be absent at any particular location. The US layer ranges in thickness from approximately 0.6 to 10 ft in this zone. The LS unit is not present in this zone due to the limited presence of the MCS layer. Glacial till is discontinuous across this zone.

The topography of the bedrock surface in this zone is accentuated by several prominent highs and one prominent valley, with up to 75 ft of relief zonewide. A relatively large and broad bedrock high extends from the BFSA north toward landfill No. 5. The bedrock consists of rocks of the Eliot Formation, which is generally composed of interbedded phyllite, metagraywacke, and quartzite.

Zone 3

Zone 3 is located in the central portion of the former AFB and occupies approximately 440 acres. The zone contains numerous buildings with adjacent paved parking areas, a network of roads, and the flight-line area. The aircraft parking apron is a major feature of the base and comprises nearly one-third of the zone. This zone contains seven IRP sites with buildings (including site 32 [with building 113]) and three underground storage tank (UST) sites. The remaining six wells are located at building 113 (site 32). Newfields Ditch, a storm-water drainage swale, passes between buildings 113 (site 32) and 119 (site 36). The ditch drains toward the northeast and eventually discharges into Hodgsons Brook.

Building 113 was used between 1955 and 1991 primarily for aircraft munitions systems and avionics maintenance (including some vapor degreasing operations). A 1200-gallon concrete UST was located near the northeastern corner of the building. The UST received waste TCE from degreasing operations conducted inside the building from 1956 to 1968. Sometime after 1977, use of the UST was discontinued and it was filled with sand. In 1988, the UST was excavated and removed, and an underground overflow discharge pipe associated with the UST was discovered. The soil and ground water contamination at this site is believed to be primarily a result of the historic use of the TCE tank and associated overflow pipe. A ground water contaminant plume of chlorinated solvents that extends beyond the source area has been delineated at this site. At this time, the ground water was to the south-southeast within this zone under static conditions (i.e., when the Haven water supply well was not being used.)

The selected remedy for this site (and site 36) included containment of the source area of dense non-aqueous liquid (DNAPL) using a vertical barrier (installed in 1996) and hydraulic control through ground water extraction and treatment (operational in February 1997 and ongoing). Containment concentration levels in the site 32 source area have consistently decreased since the implementation of this remedy (MWH Americas Inc. 2004).

The shallow subsurface beneath Zone 3 generally consists of same lithologic units found in Zone 1. The underlying bedrock is either the Kittery or Eliot formation, depending on the specific site location within Zone 3. The thickness of the overlaying unconsolidated lithologic units varies across the site because the elevation of the bedrock interface is highly variable.

Regional ground water flow is to the south-southeast within this zone under static conditions (i.e., when the Haven well is not being used). Localized flow vectors exist at each of the sites depending upon the season.

Appendix B: Tables B1–B4

Table B1. Laboratory QA/QC report.

Sample	Arsenic	Calcium	Iron	Magnesium	Manganese	Potassium	Sodium
Blank concentration (mg/L)	<0.001	<0.05	<0.05	<0.05	<0.005	<0.05	<5
LCS	1.0 (101 %R)	12 (109 %R)	10 (95 %R)	11 (104 %R)	0.90 (90 %R)	10 (94 %R)	12 (105 %R)
MS/MSD parent	<0.001	<0.05	0.18	<0.05	<0.005	<0.05	<5
Matrix spike	1.0 (104 %R)	120 (108 %R)	110 (97 %R)	120 (108 %R)	0.82 (82 %R)	110 (101 %R)	120 (106 %R)
MSD	1.1 (106 %R)	110 (103 %R)	110 (102 %R)	110 (104 %R)	0.84 (84 %R)	110 (100 %R)	120 (107 %R)
MSD RPD	2	6	5	4	2	1	1
Blank concentration (mg/L)	<0.001	<0.05	<0.05	<0.05	<0.005	<0.05	<5
LCS	1.0 (104 %R)	12 (109 %R)	11 (101 %R)	11 (98 %R)	0.91 (91 %R)	11 (102 %R)	10 (95 %R)
MS/MSD parent	0.14	61	17	12	6	1.9	110
Matrix spike	1.2 (110 %R)	180 (111 %R)	120 (96 %R)	120 (99 %R)	110 (95 %R)	100 (91 %R)	240 (117 %R)
MSD	1.1 (100 %R)	170 (101 %R)	120 (93 %R)	120 (96 %R)	94 (88 %R)	100 (90 %R)	230 (111 %R)
MSD RPD	10	9	3	3	8	1	5
Blank concentration (mg/L)	<0.001	<0.05	<0.05	<0.05	<0.005	<0.05	<5
LCS	1.0 (102 %R)	11 (100 %R)	12 (108 %R)	10 (95 %R)	1.0 (100 %R)	12 (106 %R)	11 (96 %R)
MS/MSD parent	0.15	42	11	9.3	6.5	2	100
Matrix spike	1.3 (111 %R)	150 (99 %R)	120 (103 %R)	110 (95 %R)	16 (93 %R)	100 (91 %R)	1200 (95 %R)
MSD	1.3 (111 %R)	140 (93 %R)	120 (98 %R)	110 (90 %R)	15 (87 %R)	100 (93 %R)	1200 (99 %R)
MSD RPD	0	6	5	5	7	2	4
Blank concentration (mg/L)	<0.001	<0.05	<0.05	<0.05	<0.005	<0.05	<5
LCS	1.1 (107 %R)	11 (103 %R)	12 (110 %R)	12 (108 %R)	1.1 (109 %R)	11 (102 %R)	12 (110 %R)
MS/MSD parent	0.026	62	<0.05	13	0.53	4	27
Matrix spike	1.2 (121 %R)	200 (127 %R)	110 (99 %R)	130 (110 %R)	1.5 (97 %R)	120 (107 %R)	150 (111 %R)
MSD	1.2 (115 %R)	200 (123 %R)	110 (100 %R)	130 (104 %R)	1.5 (94 %R)	120 (106 %R)	150 (109 %R)
MSD RPD	5	3	1	6	3	1	2

Table B1 (cont'd). Laboratory QA/QC report.

Sample	Arsenic	Calcium	Iron	Magnesium	Manganese	Potassium	Sodium
Blank concentration (mg/L)	<0.001	<0.05	<0.05	<0.05	<0.005	<0.05	<5
LCS	1.1 (106 %R)	11 (103 %R)	11 (99 %R)	12 (105 %R)	0.99 (99 %R)	12 (108 %R)	12 (105 %R)
MS/MSD parent	0.16	72	15	12	7.3	2.8	140
Matrix spike	1.3 (115 %R)	210 (126 %R)	120 (94 %R)	140 (119 %R)	110 (101 %R)	140 (122 %R)	280 (123 %R)
MSD	1.3 (113 %R)	210 (130 %R)	120 (94 %R)	140 (119 %R)	110 (101 %R)	130 (115 %R)	270 (113 %R)
MSD RPD	2	3	0	0	0	6	8
Blank concentration (mg/L)	<0.001	<0.05	<0.05	<0.05	<0.005	<0.05	<5
LCS	1.1 (107 %R)	11 (103 %R)	12 (110 %R)	12 (108 %R)	1.1 (109 %R)	11 (102 %R)	12 (110 %R)
MS/MSD parent	0.056	42	3.3	7.6	6.4	1.5	14
Matrix spike	1.2 (119 %R)	140 (85 %R)	120 (105 %R)	140 (120 %R)	17 (101 %R)	130 (118 %R)	150 (124 %R)
MSD	1.3 (119 %R)	150 (96 %R)	120 (103 %R)	140 (121 %R)	17 (103 %R)	130 (116 %R)	150 (123 %R)
MSD RPD	0	12	2	1	2	2	1m
Blank concentration (mg/L)	<0.001	<0.05	<0.05	<0.05	<0.005	<0.05	<5
LCS	1.1 (110 %R)	11 (104 %R)	12 (109 %R)	11 (104 %R)	1.0 (104 %R)	12 (105 %R)	12 (106 %R)
MS/MSD parent	<0.001	<0.05	<0.05	<0.05	<0.005	<0.05	<5
Matrix spike	1.1 (110 %R)	110 (96 %R)	110 (98 %R)	100 (92 %R)	1.0 (102 %R)	110 (96 %R)	100 (95 %R)
MSD	1.1 (108 %R)	110 (96 %R)	110 (97 %R)	100 (95 %R)	1.0 (104 %R)	110 (96 %R)	110 (96 %R)
MSD RPD	2	0	1	3	2	0	1

LCS, laboratory control sample; %R, percent recovery; MS, matrix spike; MSD, matrix spike duplicate; RPD, relative percent difference.

Table B2. Results for the laboratory duplicates.

		Analyte Concentration (mg/L)						
Well No.	Sample ID	Arsenic	Calcium	Iron	Magnesium	Manganese	Potassium	Sodium
32-6064 (2)	Unfiltered low-flow	0.093	110.0	1.80	38.0	0.200	9.00	53
	Lab duplicate	0.087	110.0	1.70	41.0	0.200	8.60	55
	Mean	0.090	110.0	1.75	39.5	0.200	8.80	54.0
	Standard deviation	0.0042	0.0000	0.0707	2.1213	0.0000	0.2828	1.4142
	% RSD	4.7	0.0	4.0	5.4	0.0	3.2	2.6
32-5020 (2)	Unfiltered low-flow	0.068	140.0	<0.05	38.0	0.430	7.10	57
	Lab duplicate	0.068	150.0	<0.05	43.0	0.450	7.50	66
	Mean	0.068	145.0	<0.05	40.5	0.440	7.30	61.5
	Standard deviation	0.0000	7.0711	0.0	3.5355	0.0141	0.2828	6.3640
	% RSD	0.0	4.9	0.0	8.7	3.2	3.9	10.3
13-6095	Filtered Snap Sampler	0.054	42.0	2.40	8.0	7.800	1.60	14
	Lab duplicate	0.052	41.0	2.40	7.5	7.600	1.50	14
	Mean	0.053	41.5	2.40	7.8	7.700	1.55	14.0
	Standard deviation	0.0014	0.7071	0.0000	0.3536	0.1414	0.0707	0.0000
	% RSD	2.7	1.7	0.0	4.6	1.8	4.6	0.0
13-6095	Unfiltered low-flow	0.056	42.0	3.30	7.6	6.400	1.50	14
	Lab duplicate	0.058	44.0	3.30	7.8	7.900	1.50	14
	Mean	0.057	43.0	3.30	7.7	7.150	1.50	14.0
	Standard deviation	0.0014	1.4142	0.00	0.1414	1.0607	0.00	0.00
	% RSD	2.5	3.3	0.0	1.8	14.8	0.0	0.0
32-6008	Unfiltered low-flow	0.059	98.0	0.62	34.0	0.220	10.00	95
	Lab duplicate	0.057	93.0	0.60	33.0	0.200	9.70	90
	Mean	0.058	95.5	0.61	33.5	0.210	9.85	92.5
	Standard deviation	0.0014	3.5355	0.0141	0.7071	0.0141	0.2121	3.5355
	% RSD	2.4	3.7	2.3	2.1	6.7	2.2	3.8
	Mean % RSD	2.46	2.71	1.27	4.52	5.32	2.76	3.36
	Range for % RSD	0–4.7	0–4.9	0–4.0	1.8–8.7	0–14.8	0–4.6	0–10.3
Values in parentheses indicate sampling round.								

Table B3. Results for the field duplicates.

Well No. / Sample ID	Analyte Concentration (mg/L)						
	Arsenic	Calcium	Iron	Magnesium	Manganese	Potassium	Sodium
32-6064 (1)							
Unfiltered low-flow	0.095	170	1.80	52.0	0.370	11.0	71.0
Field duplicate	0.100	180	2.10	52.0	0.400	12.0	73.0
Mean	0.098	175	1.95	52.0	0.385	11.50	72.0
Standard deviation	0.0035	7.0711	0.2121	0.0000	0.0212	0.7071	1.4142
% RSD	3.6	4.0	10.9	0.0	5.5	6.1	2.0
32-6064 (2)							
Unfiltered low-flow	0.093	110	1.80	38.0	0.200	9.00	53.0
Field duplicate	0.088	110	1.80	38.0	0.210	9.40	56.0
Mean	0.091	110	1.80	38.0	0.205	9.20	54.5
Standard deviation	0.0035	0.0000	0.0000	0.0000	0.0071	0.2828	2.1213
% RSD	3.9	0.0	0.0	0.0	3.4	3.1	3.9
32-5076							
Unfiltered low-flow	0.054	58.0	2.30	13.0	0.520	3.80	26.0
Field duplicate	0.052	62.0	2.10	13.0	0.540	4.00	27.0
Mean	0.053	60.0	2.20	13.0	0.530	3.90	26.5
Standard deviation	0.0014	2.8284	0.1414	0.0000	0.0141	0.1414	0.7071
% RSD	2.7	4.7	6.4	0.0	2.7	3.6	2.7
13-5045							
Unfiltered low-flow	0.160	72.0	15.0	12.0	7.30	2.80	140
Field duplicate	0.160	70.0	15.0	12.0	7.80	2.70	160
Mean	0.160	71.0	15.0	12.0	7.55	2.75	150.0
Standard deviation	0.0000	1.4142	0.0000	0.0000	0.3536	0.0707	14.1421
% RSD	0.0	2.0	0.0	0.0	4.7	2.6	9.4

Table B3 (cont'd). Results for the field duplicates.

	Analyte Concentration (mg/L)						
Well No. / Sample ID	Arsenic	Calcium	Iron	Magnesium	Manganese	Potassium	Sodium
13-6095							
Unfiltered low-flow	0.055	25.0	3.00	5.60	5.900	1.10	10.0
Field duplicate	0.058	29.0	3.30	6.40	5.600	1.20	10.0
Mean	0.057	27.0	3.15	6.00	5.750	1.15	10.0
Standard deviation	0.0021	2.8284	0.2121	0.5657	0.2121	0.0707	0.0000
% RSD	3.8	10.5	6.7	9.4	3.7	6.1	0.0
32-6008							
Snap Sampler	0.057	69.0	1.20	24.0	0.150	7.20	70.0
Snap duplicate	0.060	65.0	1.30	24.0	0.150	7.20	80.0
Mean	0.059	67.0	1.25	24.0	0.150	7.20	75.0
Standard deviation	0.0021	2.8284	0.0707	0.00	0.00	0.00	7.0711
% RSD	3.6	4.2	5.7	0.0	0.0	0.0	9.4
Mean % RSD for all wells	2.93	4.24	4.95	1.57	3.33	3.59	4.56
Range % RSD for all wells	0–3.9	0–10.5	0–10.9	0–9.4	0–5.5	0–6.1	0–9.4
Values in parentheses indicate sampling round.							

Table B4. Findings for the equipment blanks.

Sample ID	Concentration (mg/L)						
	Arsenic	Calcium	Iron	Magnesium	Manganese	Potassium	Sodium
Pump blank data							
Pump 1470	<0.001	<0.05	<0.05	<0.05	<0.005	<0.05	<5
Pump 2269	<0.001	<0.05	<0.05	<0.05	<0.005	<0.05	<5
Pump 2272	<0.001	<0.05	<0.05	<0.05	<0.005	<0.05	<5
Pump 1472	<0.001	<0.05	<0.05	<0.05	<0.005	<0.05	<5
Pump 2292	<0.001	<0.05	**0.14**	<0.05	**0.006**	<0.05	<5
Pump 2271	<0.001	<0.05	**0.27**	<0.05	<0.005	<0.05	<5
Pump 2291	<0.001	<0.05	<0.05	<0.05	<0.005	<0.05	<5
Pump 2116	<0.001	<0.05	<0.05	<0.05	<0.005	<0.05	<5
Pump 2270	<0.001	<0.05	<0.05	<0.05	<0.005	<0.05	<5
Pump 1400	<0.001	<0.05	**0.18**	<0.05	<0.005	<0.05	<5
Pipette cleaner	<0.001	<0.05	**0.09**	<0.05	<0.005	<0.05	<5
Lab Filter Blank	<0.001	<0.05	<0.05	<0.05	<0.005	**0.06**	<5
Snap Sampler and RGC sampler blank data							
RGC-unfiltered	<0.001	**0.13**	<0.05	<0.05	<0.005	**0.07**	<5
SS-unfiltered	<0.001	<0.05	<0.05	<0.05	<0.005	<0.05	<5
SS-filtered	<0.001	<0.05	**0.05**	<0.05	<0.005	<0.05	<5
PC-unfiltered	<0.001	<0.05	<0.05	<0.05	<0.005	<0.05	<5
PC-unfiltered Dupe	<0.001	<0.05	<0.05	<0.05	<0.005	<0.05	<5
PC-filtered	<0.001	<0.05	**0.19**	<0.05	<0.005	<0.05	<5
Filter blank	<0.001	<0.05	<0.05	<0.05	<0.005	<0.05	<5
Lab filter blank	<0.001	<0.05	<0.05	<0.05	<0.005	<0.05	<5

Detection levels are shown for all nondetects.
Bolded numbers indicate analyte concentrations above the detection limit.
PC, pipette cleaner.

REPORT DOCUMENTATION PAGE	*Form Approved* *OMB No. 0704-0188*

1. REPORT DATE *(DD-MM-YYYY)* December 2008	**2. REPORT TYPE** Technical Report	**3. DATES COVERED** *(From - To)*

4. TITLE AND SUBTITLE Evaluation of the Snap Sampler for Sampling Ground Water Monitoring Wells for Inorganic Analytes	**5a. CONTRACT NUMBER** **5b. GRANT NUMBER** **5c. PROGRAM ELEMENT NUMBER**
6. AUTHOR(S) Louise V. Parker, Nathan D. Mulherin, and Gordon E. Gooch	**5d. PROJECT NUMBER** **5e. TASK NUMBER** **5f. WORK UNIT NUMBER**
7. PERFORMING ORGANIZATION NAME(S) AND ADDRESS(ES) U.S. Army Engineer Research and Development Center Cold Regions Research and Engineering Laboratory 72 Lyme Road Hanover, NH 03755-1290	**8. PERFORMING ORGANIZATION REPORT NUMBER** ERDC/CRREL TR-08-25
9. SPONSORING / MONITORING AGENCY NAME(S) AND ADDRESS(ES)	**10. SPONSOR/MONITOR'S ACRONYM(S)** **11. SPONSOR/MONITOR'S REPORT NUMBER(S)**

12. DISTRIBUTION / AVAILABILITY STATEMENT
Approved for public release; distribution is unlimited.

13. SUPPLEMENTARY NOTES

14. ABSTRACT

Laboratory studies and a field demonstration were conducted to determine the ability of the Snap Sampler to recover representative concentrations of inorganic analytes, including metals, from ground water. For the laboratory studies, concentrations of analytes in Snap Sampler samples were compared with concentrations of the analytes in samples collected from a standpipe (i.e., control samples). These studies clearly demonstrated that Snap Sampler concentrations were comparable to control sample concentrations.

For the field demonstration, 10 sampling events were conducted at the former Pease Air Force Base. Samples taken using a Snap Sampler were compared with samples collected using conventional low-flow purging and sampling and a regenerated cellulose passive diffusion sampler. Analytes included calcium, iron, magnesium, manganese, potassium, sodium, and arsenic. With one exception, Snap Sampler concentrations were equivalent to those in low-flow samples. Concentrations of unfiltered iron were higher in the Snap Sampler, especially in the two stainless steel wells, and reflected turbidity levels in the wells at the time of sampling. Elevated turbidities may have resulted from installing additional sampling equipment (including the baffle, pump, samplers, and bottom weight) in the well before sampling. We will examine this issue further at our next test site.

15. SUBJECT TERMS

dialysis membrane sampler, equilibrated-grab samplers, HydraSleeve sampler, low-flow purging and sampling, no-purge sampling, passive ground water sampling, passive sampling, regenerated cellulose sampler, Snap Sampler

16. SECURITY CLASSIFICATION OF:			**17. LIMITATION OF ABSTRACT**	**18. NUMBER OF PAGES**	**19a. NAME OF RESPONSIBLE PERSON**
a. REPORT	**b. ABSTRACT**	**c. THIS PAGE**			**19b. TELEPHONE NUMBER** *(include area code)*
U	U	U	U	72	

DEPARTMENT OF THE ARMY
ENGINEER RESEARCH AND DEVELOPMENT CENTER, CORPS OF ENGINEERS
COLD REGIONS RESEARCH AND ENGINEERING LABORATORY, 72 LYME ROAD
HANOVER, NEW HAMPSHIRE 03755-1290

Official Business