Technical Guidance Manual for Hydrogeologic Investigations and Ground Water Monitoring

Chapter 10
Ground Water Sampling

Mike DeWine, Governor
Jon Husted, Lt. Governor
Laurie Stevenson, Director

October 2020

Technical Guidance Manual
for Hydrogeologic Investigations
and Ground Water Monitoring

Chapter 10
Ground Water Sampling

October 2020

Revision 3

Ohio Environmental Protection Agency

**Division of Environmental Response and Revitalization
in collaboration with Division of Drinking and Ground Waters
and Division of Materials and Waste Management**

**P.O. Box 1049
50 West Town Street
Columbus, Ohio 43216-1049
Phone: (614) 644-2924**
epa.ohio.gov/derr/

Preface

This document is part of a series of chapters incorporated in Ohio EPA's *Technical Guidance Manual for Hydrogeologic Investigations and Ground Water Monitoring* (TGM), which was originally published in 1995. DERR now maintains this technical guidance in collaboration with DDAGW and DMWM as a series of chapters rather than as an individual manual. These chapters can be obtained at *epa.ohio.gov/derr/gw_support#184203976-technical-guidance-manual-tgm.*

The TGM identifies technical considerations for performing hydrogeologic investigations and ground water monitoring at potential or known ground water pollution sources. The purpose of the guidance is to enhance consistency within the Agency and inform the regulated community of the Agency's technical recommendations and the basis for them.

Ohio EPA utilizes **guidance** to aid regulators and the regulated community in meeting laws, rules, regulations and policy. Guidance outlines recommended practices and explains their rationale. The methods and practices described in this guidance are not intended to be the only methods and practices available to an entity for complying with a specific rule. Unless following the guidance is specifically required within a rule, the Agency cannot require an entity to follow methods recommended within the guidance. The procedures used should be tailored to the specific needs and circumstances of the individual site, project and applicable regulatory program, and should not comprise a rigid step-by-step approach utilized in all situations.

Changes from the May 2012 Manual

The Ohio EPA Technical Guidance Manual for Hydrogeologic Investigations and Ground Water Monitoring (TGM) was first finalized in 1995 and subsequently updated in February 2006 (Revision 1) and May 2012 (Revision 2). This guidance document represents an update (Revision 3) to Chapter 10 (Ground Water Sampling). Listed below are the technical changes from the 2012 version of Chapter 10.

1. Updated formula for calculating water volume in a one-foot section of well casing and associated table.

2. Updated sampling mechanisms and associated table.

3. Added general information related to sampling PFAS.

4. Updated Field Conditions section for additional conditions that could affect the quality of the .sample.

5. Updated water level measurements section and associated table.

6. Updated reference list.

Table of Contents

Chapter 10
Ground Water Sampling

This chapter summarizes procedures for collecting ground water samples from monitoring wells. It focuses on the planning and preparation prior to sampling, types of sampling and purging equipment, field procedures, quality control sampling, and documentation to ensure that samples represent the quality of water obtained from the sampled interval. When selecting protocol, it is important to understand the impacts that removing water from a well can have on the chemistry of the water. Therefore, impacts to sample integrity are also discussed. The chapter also provides some information on the selection of analytical methods and laboratory quality assurance. Understanding the purpose of sampling with respect to the regulatory program requirements and planning in advance is very important. In addition, site location, age, other potential sources, the proximity of water supply wells, and sampling for the analysis of additional constituents may be useful. Consistent sampling and analytical methodology are integral to quality monitoring data.

The primary objective of most ground water monitoring programs is to collect a sample that represents the in-situ ground water quality. However, the working definition of "representative" is not always the same for all programs. For example, those interested in characterizing ground water for the purpose of evaluating it as a potable water supply may be more interested in volumetric-averaged concentrations in the ground water zone (Nielsen and Nielsen, 2006). Monitoring programs may also be designed to determine "worst-case" conditions. Therefore, prior to starting any monitoring program, the data quality objectives need to be determined to ensure the collection of data that are of adequate quality to support decision making (See U.S. EPA, 2006).

The goal in sample collection is to sample in a manner that results in the least disturbance or change in the chemical and physical properties of the water. The guidelines provided here are intended to assist in choosing the most appropriate methods. Site-specific circumstances may require alternative approaches that are not specified. In these cases, the appropriate regulatory authority should be contacted to establish an acceptable approach. In addition, rules may specify issues such as frequency of sample collection, filtration, frequency and accuracy of water level measurements, and parameters for analysis. Requirements for documentation of field and laboratory procedures may also be specified. Appropriate divisions within Ohio EPA should be consulted when planning a ground water sampling program.

The choice of equipment and methodology should be based on an understanding of the hydrogeology of the area and the purpose of the data collection. Each technique has disadvantages and advantages. Because different techniques may yield different results, the best approach is to be consistent throughout an investigation to facilitate the comparison of data values over time (ASTM D4448). When necessary, changes in sampling strategies should be discussed with Ohio EPA prior to implementation.

Although the chapter is intended specifically for the sampling of conventional monitoring wells, the procedures may be useful for other types of ground water sampling, such as direct push technology and water supply wells. Additional information on direct push can be found in Chapter 15-Use of Direct Push Technologies for Soil and Ground Water Sampling. Additional considerations for sampling a water supply well can be found in Appendix A of this chapter.

1.0 Factors Influencing Sample Quality

Many aspects of the sampling process can affect the chemistry and physical properties of ground water when it is being collected. As a result, a sample may not represent the actual quality of the ground water. Therefore, the potential effects need to be considered in any sampling program.

1.1 Well Construction and Development Concerns

1.1.1 Well Construction

A detailed understanding of the local hydrogeology is critical for proper monitoring well construction. Wells must be properly placed with respect to the targeted saturated zone, grouted and sealed. The screen must not interconnect multiple saturated zones. Otherwise, the chemistry and physical properties of a ground water sample may be affected by poor well construction. Ground water samples from wells that have inadequate filter packs, are improperly grouted, or have misplaced screens may have ground water flowing through their intakes that is not represent the ground water quality of the saturated zone. This may be due to grout contamination, water seeping down the casing, or mixing from the surface or other ground water zones.

1.1.2 Well Development

Prior to sampling the well for the first time, care should be taken to properly develop the well. Proper well development creates a graded filter pack around the well screen. It reduces the sample turbidity by removing the fine particulate matter. If a well has not been properly developed, then sample quality may be affected by the sediments in the water. If the well begins to silt up over time, the well may need to be redeveloped. Chapter 8 - Monitoring Well Development, Maintenance, and Redevelopment, discusses proper development techniques and when a well can be considered properly developed.

1.1.2.1 Time Interval Between Well Installation and Well Development.

Development should not be implemented until the seal has cured and settled to prevent pulling uncured grout into the sand pack. The time interval between well installation and development is a function of well construction, type of grout, and conditions under which the grout was installed. For example, neat cement (Type 1) generally cures within 48 hours (Gaber and Fisher, 1988, State Coordinating Committee on Ground Water, 2015); however, if the cement grout column is located mostly above the water column, then 24 hours may be sufficient. Bentonite-based grouts tend to set within 24 to 48 hours. Bentonite granules, chips or pellets, above the sand pack will reduce the potential of contaminating the sand pack with grout during development activities; thus, reduce the time interval between construction and development. A 3/4-inch direct push well installed with dry granular bentonite grout may require less than 24 hours

Ideally, a time of 48 hours is recommended (U.S. EPA, 2001b). However, shorter time frames may be acceptable as discussed above. If shorter time frame is used, then justification should be documented. It is ultimately the responsibility of the environmental professional to demonstrate that the ground water samples collected from the well are representative of the formation water and are not impacted by grout contamination.

1.1.2.2 Time Interval Between Well Development and Sampling

Prior to sampling a well, sufficient time should be allowed for equilibration with the formation after development. The time interval between well development and sampling is dependent on physical equilibrium (i.e., return to static water level conditions), chemical equilibrium between the well and saturated zone, well drilling and construction, and data quality objectives.

Physical equilibrium (return to static water level conditions) is verified by measuring the water level in the well. Depending on the well diameter and saturated zone characteristics, the time required to return to physical equilibrium could range from several minutes to several days to over a month.

Evaluating chemical equilibrium is a more complicated matter. A simple method for determining when a monitoring well and the surrounding formation materials are in chemical equilibrium is not currently available. The degree to which a well disturbs the saturated zone chemistry depends on the drilling method, installation technique, construction, development technique and hydrogeologic environment (Striggow et al., 2008; Kim 2003; Lapham et al. 1995). All of these factors influence the amount of time needed for well materials to achieve chemical equilibrium with the surrounding formation materials and ground water. For example, a 2-inch monitoring well installed in bedrock using air rotary would require more time to chemically stabilize after development than a ¾-inch direct-push well with a pre-packed screen, because installation of the 2-inch

monitoring well causes more formation damage and occupies a greater volume of the saturated zone. Several days may be required for the 2-inch well to stabilize with the surrounding formation, whereas the direct-push well may stabilize within a day.

Ohio EPA recommends that ground water samples not be collected from a newly-developed (or redeveloped) well until the well has physically stabilized, i.e., returned to static water level conditions. In addition, ground water sampling should be delayed until the well materials have had sufficient time to achieve chemical equilibrium with the surrounding saturated zone. While rigorous scientific techniques are not currently available to substantiate such a time frame (U.S. Army Corps of Engineers 1998), consideration should be given to the drilling method, installation technique, construction, development technique and hydrogeologic environment (Striggow et al., 2008; Kim 2003; Puls and Barcelona 1996; Lapham et al. 1995; Aller et al., 1991). For shallow (< 25 ft), small diameter (< 2 inch) monitoring wells installed using direct push methods, Ohio EPA recommends a post-development stabilization period of at least 24 hours prior to sampling. For deeper (≥ 25 ft) direct-push wells or deeper (25 to 50 ft), larger diameter (2 to 4 inch) "traditional" monitoring wells installed in unconsolidated sediments using augers or rotary drilling methods, Ohio EPA recommends a post-development stabilization period of at least 72 hours prior to sampling. For deep (> 50 ft) traditional monitoring wells installed in unconsolidated sediments or installed in bedrock, Ohio EPA recommends a post-development stabilization period of at least one week (7 days) prior to sampling. Monitoring wells installed in low permeability silt, clay or shale will generally require a longer stabilization period than wells installed in sand or gravels or more permeable bedrock types (U.S. Army Corps of Engineers 1998; Izraeli et al. 1992.)

1.2 Effects Caused by Change in Sample Environment

Transfer of ground water from *in-situ* to atmospheric conditions can alter its chemistry significantly unless proper sampling techniques are used. Aeration/oxidation, pressure, and temperature changes are three major causes of chemical alteration.

1.2.1 Aeration/Oxidation

Upon exposure to the atmosphere, the oxidation state of ground water samples can increase due to the addition of oxygen. Dissolved species such as iron (Fe), manganese (Mn), arsenic (As), and cadmium (Cd) may be oxidized from a reduced state (Gillham et al., 1983), which can cause them to precipitate from solution. The oxidation of Fe is particularly important for sample stability. Ground water may contain high concentrations of dissolved Fe due to anoxic (low oxygen) subsurface conditions. Upon exposure, it can oxidize rapidly and precipitate ferric hydroxide, resulting in a decrease in pH that may alter sample integrity further $[2Fe^{3+} + 6H_2O \rightarrow 2Fe(OH)_{3(s)} + 6H^+]$. Ferric hydroxide is known to remove contaminants from solution including, but not limited to, copper (Cu), zinc (Zn), cobalt (Co), cadmium (Cd), arsenic (As) and lead (Pb). Due to oxidation other metals may be dissolved or desorbed and become mobile. While constituents such as Pb do not necessarily change oxidation state, if the solid Pb is adsorbed to changes oxidation state and is dissolved, it releases Pb and other absorbed metals/constituents/trace elements. Alternatively, when oxidized other metals tend to dissolve and become mobile, such as chromium (Cr)...While it may often be difficult to prevent redox changes, acidification, and refrigeration of samples being analyzed for metals should prevent metals from precipitating or reacting to alter oxidation state.

1.2.2 Pressure Differences

Pressure changes caused by the release of ground water into a well may cause shifts in chemical equilibrium (US EPA, 2002) Ground water may have high partial pressures of carbon dioxide (CO_2) gas and, upon exposure, degas CO_2. This is known to cause increases in pH by up to 0.5 to 1 Standard Units (S.U.) and may cause various metals to dissolve or precipitate. If volatile organic compounds (VOCs) are present, sudden pressure changes can cause their volatilization. This will result in a negative bias with respect to true VOC concentration.

1.2.3 Temperature Differences

The temperature of a sample may change because of differences between ambient air and subsurface conditions. A primary concern is an increase in temperature, which may kinetically favor redox reactions and promote increased biodegradation and volatilization (US EPA, 2002).

1.3 Effect Due to Sampling Technique

The method and design of the sampling device potentially can alter samples (US EPA, 2002). Tools that allow air to contact ground water (see Section 3.0-Sampling and Purging Equipment) can potentially aerate samples, as discussed above. Devices can leach contaminants into samples or sorb contaminants from them. Also, improper decontamination of equipment can alter samples.

2.0 Planning and Preparation

The success of any ground water sampling event hinges on the planning and preparation conducted prior to entering the field. The sampling procedures should be documented in a written plan. Items that should be included in the written plan are summarized below. Procedures and event planning and preparation should be evaluated carefully and be appropriate for the associated Ohio EPA program and the intended use of the sampling data. This should also include an evaluation of the analytes selected.

2.1 Written Plan

Written, detailed, site-specific protocol should be developed to document sampling and analysis procedures. The protocol can be incorporated into a single, stand-alone document (sometimes called a sampling and analysis plan) or can comprise a section of a more comprehensive document. Protocol should provide sufficient detail for personnel to properly operate equipment and perform procedures and techniques in a manner that will generate representative data. The circumstances and conditions under which procedures and techniques will be implemented should be clearly described.

The submittal, format, and/or disposition may or may not be specified by rule. In all cases, the plan or other protocol should meet all requirements of the associated Ohio EPA program and provide data that are appropriate for the investigative purposes and that meet data quality objectives. In general, a plan may include (at a minimum) the components listed in Table 10.1.

Table 10.1 *GENERAL COMPONENTS[1] OF A GROUND WATER SAMPLING AND ANALYSIS PLAN (SAP)*		
Parameter selection • organics • metals	**Sampling frequency** • quarterly, semi-annual, annual • quality control samples	**Decontamination procedures** • equipment • IDW management
Field procedures prior to sampling ground water: • well inspection • water level measurements (including meter type and level of accuracy) • total depth of well • detection and sampling of immiscible liquids • field analyses • purging of well	**Well purging, including but not limited to:** • methods • criteria completion • equipment • disposal of water **Sample withdrawal:** • methods • equipment	**Field measurements of ground water:** • parameters (e.g., pH, temperature, and conductivity) • description and calibration of field equipment • description of field analysis procedures
Sample handling: • order of collection • filtration[2] • preservation (type and when/how added) • containers with labels • holding times • shipping	**Documentation:** • field logbook or sampling documentation forms[3] • standardized chain-of-custody forms • sample analysis request sheet • field QA/QC samples	**Laboratory analysis:** • analytical methods • detection limits • laboratory QA/QC samples • description of data validation methods reporting requirements and format

2.2 Sampling Event Planning and Preparation

Before any sampling begins, planning and preparation should be a high priority. All personnel should be familiar with site-specific written protocol and trained in the proper use of the equipment. All equipment and paperwork should be organized. Instruments should be in working order, properly decontaminated, and calibrated. Field logs, sheets, or other documents used to record notes should be organized. Arrangements with the laboratory

---

[1] Additional components may be necessary on a site-by-site basis.

[2] Check whether the regulatory program allows filtering of ground water samples. Note that the Ohio solid waste regulations do not allow filtering at municipal landfills.

[3] See Section 4.10 - Documentation for items that may need to be included.

should be made to ensure that samples can be handled and analyzed within the required holding times and to obtain labels, appropriate containers, and preservatives. The following are general checklists for preparation procedures and equipment:

2.2.1 Preparation Procedures
Determine the following:

- Sampling date, time, and location
- Total sampling and travel time to insure appropriate lab arrangements
- Number and type of analyses needed from each location
- Purge water management practices
- Decontamination procedures
- Safety procedures
- Number of field blanks, equipment rinsates, trip blanks and duplicates needed
- Sample volumes needed, total number of samples, and container type
- Samples to be filtered (if appropriate)
- Appropriate equipment
- All equipment is properly calibrated and in working order
- Bailer volume (if necessary)
- Number and type of containers needed
- Number and type of necessary preservatives
- All chain-of-custody procedures

In addition, Review the construction, sampling history and recharge rate of each well. Be aware of any nearby production wells that may affect measured water levels

2.2.2 General Supply and Equipment Checklist

- SAP
- Keys to locks on wells
- Map of site, well locations, and underlying geology
- Field notebook, logbook, and/or field sampling forms
- Indelible marking pens
- Appropriate lab analysis and chain-of-custody forms
- Preservatives
- Filtration equipment
- Ice
- Coolers for ice and samples
- Purging and sampling devices
- Appropriate tubing
- Appropriate sample containers and labels
- Field monitoring meters (e.g., water level, pH, specific conductance, temperature)
- Calibration instructions and standard testing solutions for field monitoring equipment
- Calibrated bucket (to determine volume of purged water)
- Tool box
- Extra batteries
- Safety equipment
- Calculator
- Plastic sheeting
- Decontamination solutions and equipment
- Flashlight
- Photoionization detector (PID) or organic vapor analyzer (OVA)
- Equipment for detecting immiscible fluids (e.g., interface probe or clear bailer)
- Contact information for site, facility, and laboratory
- Emergency contact information

2.3 Sampling Frequency
For sites that require a ground water monitoring program, regulations may specify sampling frequencies. For other cases where contaminants have been identified and no regulatory requirements have been identified, the following hydrogeologic considerations in determining sampling frequencies are provided. Frequencies may be determined by site hydrogeology. U.S. EPA (1992) suggested using the Darcy equation to determine average linear velocity of ground water (see Chapter 3). If velocity is relatively low, less frequent sampling may be required. High velocity necessitates the need for more frequent sampling. Linear flow velocity can be used to

determine an interval that yields an independent sample. Sample independence is an important concept for statistical and analytical data analysis (U.S. EPA, 2009). Barcelona et al. (1985) provided a graph based on hydraulic parameters that can be useful. Barcelona et al. (1989) indicated that data collected over a period of two years or more is often needed to establish seasonal trends before an adequate frequency can be selected.

The type of contaminant source (i.e., spill, intermittent source, or continuous source) should also be considered for sites with releases and known contamination (Barcelona et al., 1985). Spills tend to move as a slug/plume through the subsurface, potentially limiting the sampling time frame at a particular well as the slug passes. Intermittent sources may produce slugs with high and low concentration trends to develop. Continuous sources may produce a large plume requiring a sampling frequency based on ground water flow velocity.

2.4 Parameter Selection

Parameter selection depends on whether the purpose of sampling is to 1.) quantify the general quality of the ground water; 2.) identify the presence of any contamination, or 3.) identify site as the specific source of contamination.

2.4.1 Parameters to Characterize General Quality

Parameters used to characterize general quality can include: pH, alkalinity, total dissolved solids (TDS), turbidity, dissolved oxygen (DO), oxidation/ reduction potential (ORP), fluoride (F^-), hydrogen sulfide (H_2S), total hardness and non-carbonate hardness, specific conductance, chloride (Cl^{-1}), nitrate (NO_3^{-1}), sulfate (SO_4^{-2}), phosphate (PO_4^{-3}), silicate (SiO_2), sodium (Na^{+1}), potassium (K^{+1}), calcium (Ca^{+2}), magnesium (Mg^{+2}), ammonium (NH_4^{+1}), total iron (Fe), and manganese (Mn). The results can provide an overall picture of ground water geochemistry that is useful to site characterization. For example, an understanding of geochemistry can help in determining chemical species present (e.g., AsO_3^{-2} vs. AsO_4^{-3}) and mobility in the subsurface. Certain parameters (e.g., anions, cations, pH, TDS, specific conductance) are helpful in evaluating releases of inorganic contaminants, while other parameters (chloride, iron, nitrate, sulfate, DO, ORP, and alkalinity) can be used to evaluate changes in ground water chemistry caused by the release and biodegradation of organic contaminants. Regulated entities (such as municipal or hazardous waste landfills) may be required to establish a sampling program that includes some of the above-mentioned parameters.

2.4.2. Parameters to Characterize Contamination

When ground water contamination is known, suspected, or being investigated as part of a monitoring program, parameters specific to the waste material, history of the site/facility, or chemicals of concern (COCs) usually are necessary. Rules may also dictate specific parameters. When ground water contamination is known or suspected, entities may be required to monitor additional site-specific parameters[4].

Past waste constituents, handling practices, and age of site should be considered. Because waste released to the environment may chemically change through time, potential breakdown products should be considered. If accurate disposal records are available and waste constituents are well documented, the list of parameters can be relatively limited. The list should be more extensive if handling practices are poorly understood, or if needed to differentiate source. In more industrial areas, source of contamination may not be evident using the common contaminant criteria alone and require advanced techniques/analysis such as multivariate statistics, principle component analysis, hierarchical cluster analysis, time series modeling, and GIS analysis techniques. If this is the case, parameters should also be consistent with other regulated sites that may be contaminated the same area/region in order to properly compare sampled groundwater and identify contaminating source. Monitoring suites of parameters (e.g., volatiles, semi-volatiles, metals, etc.) may be necessary when specific waste constituents are not known. Where rule/policy allows, lists may be narrowed as the investigation progresses and waste constituents and chemicals of concern become better defined.

---

[4] Ohio EPA recommends that, in some cases (e.g., characterizing known ground water contamination), that the laboratory be requested to report all constituents listed in a method's target analyte list whether they are detected or quantified or not. This ensures that breakdown products are also considered.

3.0 Sampling and Purging Equipment

Site-specific hydrogeology, geochemistry, types of contaminants, and well design determines the specific equipment recommended. Ultimately, the chosen equipment should employ inert material, not subject samples to negative pressures or high positive pressures and minimize exposure of samples to the atmosphere (ASTM D4448). The following are specific criteria and characteristics for equipment/device selection.

3.1 Criteria for Selection

In general, the choice of a device should be based on the ***characteristics of the device*** in combination with the ***characteristics of the site/project*** as listed in the following sections.

3.1.1. Device Characteristics

Characteristics of devices are:

- ***Device composition*** - The chosen device should have sample-contacting parts made of "inert" materials that limit the potential for bias through sorption or leaching of contaminants, degradation, or corrosion. For components requiring rigid material (casing, screen, bailers etc.), the acceptable materials are fluorocarbon polymer (e.g., Teflon®), stainless steel (316 and 304), and polyvinyl chloride (PVC). Disposable bailers can also be composed of polyethylene and polypropylene. When sampling for organics, pump tubing should be composed of fluorocarbon polymer, or fluorocarbon polymer-lined polyethene. Polyethene tubing is also acceptable for sampling for inorganics (USGS, 2014; Yeskis and Zavala, 2002, ASTM D4088). Refer to Section 3.2 for types of equipment and the limitations of their use based on material composition when sampling for organic/inorganic constituents. Special material and design considerations are needed to ensure compatible materials are used when sampling for Per- and Polyfluoroalkyl Substances (PFAS), as discussed in Section 3.2.10 below.

- ***Device design and technique of use*** - The device should deliver samples with minimal atmospheric exposure, should not apply negative pressures (vacuum), and should limit agitation, both in the well and in the transfer process (ASTM D6634). Furthermore, the tool should not introduce air or non-inert gas into samples as part of its lift mechanism.

- ***Flow rate control and capacity*** - When pumps are used, low flow rates are desirable to limit agitation and turbulent flow, especially for VOCs (Barcelona et al., 1985, U.S. EPA, 1986a). The ability to maintain a steady low flow varies significantly. If the device is being used for purging and sampling, then it should be capable of being operated at variable flow rates suitable for both applications. Flow control that involves "valving" should be avoided, since it can cause pressure changes and subsequent sample alteration. Instead, a mechanism that directly controls the rate (i.e., a rheostat to vary the power supplied to an electric submersible pump) should be utilized.

- ***Operation and Maintenance*** - The device should be easy to operate and maintain. If personnel are not properly trained, the margin of potential error is greater. The device should be designed for in-field maintenance. Mechanically simple equipment that can be easily repaired with inexpensive, replaceable parts is preferable. If decontamination is necessary, the device should be easy to decontaminate. Devices that are constructed to minimize the surface area contacting ground water samples and that are easy to disassemble and reassemble are best. Use of dedicated or disposable equipment at each well or sampling point eliminates the need for decontamination, saving valuable field time and reducing the potential for cross contamination of samples.

- ***Device reliability, durability, and portability*** - The device should operate reliably for extended periods and be able to withstand a variety of chemical and physical environments. Dedicated equipment may need to withstand extended contact with ground water and any existing contamination. Equipment that is transported into locations where access is limited should be sufficiently portable. Excess weight and volume of battery packs, generators, air compressors, tubing, etc. can limit portability.

- ***Capital, operation, and maintenance costs*** - These should be considered however, they should not

be overriding factors. Obtaining a sample that is representative of site conditions should be of more importance than cost, particularly when the costs of well installation, chemical analysis, and possible litigation resulting from discrepant analytical results are considered. These costs often far outweigh equipment purchase costs (Nielsen and Yeates, 1985).

3.1.2 Site/Project Characteristics
Characteristics of sites/projects that should be considered are:

- **Project and data quality objectives –** What is the purpose of the sampling? Are samples being collected for compliance evaluation, risk assessment or field screening?

- *Monitoring Well Diameter* - The device should be compatible with the diameter of the well. Most sampling equipment is not designed to be used in all wells.

- *Well Obstructions or Constrictions* - These can hinder the entry and retrieval of sampling equipment. For example, casing joints may not be flush and could prevent insertion. Also, a well that is not plumb can restrict access.   For example, a well installed in an area with slope stability issues can be crimped, bent, or broken from the slope movement.

- *Depth to the Sampling Interval* – As well depth increases, greater lift capacity is required and sampling time generally increases, which may limit the desirability of labor-intensive devices. Options generally become limited as depth increases.

- *Parameters of Interest* - The suitability of various devices may depend on the parameters of greatest concern. Some devices perform better for inorganics, while some are more suitable for VOCs. Some materials contained within or frequently used with ground water sampling equipment are not suitable for sampling for PFAS (refer to Section 3.2.1).

- *Presence of Non-Aqueous Phase Liquids (NAPLs)* - The equipment should be capable of detecting the presence of either light or dense NAPLS if they are potentially present.

- *Saturated Zone Characteristics and Ground Water Chemistry* - The equipment should be appropriate for the saturated zone yield, the screen or open borehole length, the presence of stratification (causing vertical variation in yield) within the screened saturated zones, and the available water column in the well. Additionally, the sampling equipment should be compatible with ambient ground water chemistry, unusually low (<5.5.U.) or high (>9.5.U.) pH conditions, the presence of gas, etc.

- *Temporal (Seasonal) Variations* - The sampling equipment should be operable over seasonal variations in saturated zone temperature, yield and water level elevation.

3.2 Types of Equipment
The following is a discussion of commonly used sampling equipment available. Table 10.3 (see end of Section 3.2.10) summarizes the recommended devices. Devices not mentioned may be acceptable if they are peer-reviewed and have been demonstrated to be capable of collecting representative samples. For additional information, see ASTM D4448, ASTM D6634, Barcelona et al. (1985), Nielsen and Yeates (1985), Electric Power Research Institute (EPRI, 1985, 1987), Gillham et al. (1983), Nielsen and Nielsen (2006), KEMI (2015), Parker (1994), Pohlman and Hess (1988), U.S. EPA (1992), and Yeskis and Zavala (2002).

3.2.1 Grab Samplers
Grab samplers collect a sample at discrete depths without being pumped or lifted to the surface by gas or air. Grab samplers are commonly used to collect ground water include bailers and syringe samplers.

3.2.1.1 Bailers
Bailers are the most portable of all sampling devices. A bailer can be constructed of virtually any rigid or flexible material, including materials that are inert to chemical contaminants. For sampling ground water, acceptable compositions include Teflon®, stainless steel, PVC, polyethylene, and polypropylene. The cord used to raise and lower the bailer should be made of a non-reactive substance (e.g., Teflon-coated wire/rope, polypropylene).

Bailers are readily available in a variety of diameters. Their diameter should be 75% (or less) of the inside

diameter of the well casing to allow for adequate clearance. There are several types of bailers (ASTM D6634, ASTM D6699):

- A ***top filling*** bailer is designed such that water flows through its top. Because of the agitation of the sample, this bailer is only appropriate for sampling light, non-aqueous phase liquids (LNAPL).

- A ***single check*** valve bailer (open bailer) has a valve at its bottom that seals the sample chamber when the bailer is withdrawn.

- A ***double check*** valve bailer (point source bailer) is designed to sample discrete zones in a water column. Water flows through valves at both ends as the bailer is lowered. When the desired level is reached, the bailer is pulled back, both valves close, and water from the interval is retained. However, if appropriate procedures are not carefully followed, samples collected may not be representative of the depth interval of interest. The double check valve bailer is also effective in collecting dense, non-aqueous phase liquids (DNAPLs).

- A **differential pressure** bailer consists of a sealed canister body with two small diameter tubes of different heights. The bailer is rapidly lowered into the well. When the descent has stopped, differences in hydrostatic pressure between the two tubes allow the bailer to fill through the lower tube as air is displaced through the upper tube. This minimizes the exposure of the sample to air, especially if the bailer is fitted with internal 40-millimeter vials for direct sample bottle filling. However, because the bailer is lowered rapidly, it will agitate the water column.

Some studies have demonstrated that levels of VOCs in samples obtained with bailers are statistically lower than in samples obtained with other devices (Imbrigiotta et al. 1988; Tai et al. 1991). In addition, bailing can cause increased turbidity (Puls and Powell, 1992; Puls et al., 1992; Backhus et al., 1993). In contrast, a literature survey by Parker (1994) found that bailers can recover representative samples under certain circumstances and that loss of volatile and oxidizable analytes can be reduced by careful use of bottom-emptying devices and a study comparing results from a bottom-emptying bailer and a Keck® helical-rotor pump operated at low flow pumping rates determined that differences in VOC concentrations were relatively small (Karkins, 1996).

Bailers may not be the best available technology. However, they may be the only practicable option for sampling some ground water zones. Bailers may be preferred where the water column is shallow, or the saturated zone is very deep. They may be preferred when concentrations of contaminants are extremely high because they are easier to decontaminate and are less expensive to replace than pumps. Disposable bailers eliminate the need to decontaminate.

Rapid addition or withdrawal of a bailer can cause surging within the well that may cause increased turbidity, loss of volatiles, aeration, degassing of samples, and affect the level of development of the well. Personnel sampling with bailers need to be properly trained since the results are highly dependent on the skill, care, and consistency of the operator. This training should be documented in the SAP. If samples from bailing are turbid, then it should be determined whether sampling with a low flow method would produce less turbid samples.

When bailing, ***double check valve bottom-draining*** bailers are recommended. This allows for lessened sample disturbance during transfer to the container. The bailer should be composed of Teflon®, stainless steel, PVC, polyethylene, or polypropylene. Either fluorocarbon polymer-coated or colorless (white) polypropylene cord should be used to lower and raise the bailer. Polypropylene cord is inexpensive enough to be discarded after one use. A bailer should always be lowered and raised slowly to minimize sample agitation associated with degassing, aeration, and turbidity and to the extent possible, avoid hitting the sides of the well. A tripod and pulley may be used to remove the bailer.

Pouring water from the top of a bailer directly into a container or to a transfer vessel may agitate/aerate the sample and alter its chemistry; therefore, such pouring is discouraged.

3.2.1.2 Syringe Samplers
Syringe samplers may be used for low-volume sampling for inorganics and non-volatile organics. These samplers can operate at great depths to provide discrete samples from specific intervals or zones. A sample container is pressurized or evacuated and lowered into a well. The sample is collected by opening the container

or releasing the pressure, drawing water into the sampler (Nielsen and Nielsen, 2006). The syringe sampler is withdrawn, and the sample is transferred to a collection bottle, or alternatively, the syringe sampler can be utilized as the sample container.

Syringe devices cannot be used for purging large volumes and are ineffective for collecting large samples. In addition, ground water containing high concentrations of suspended solids may cause the syringe device to leak (U.S. EPA, 1992). Researchers have concluded that these samplers are inferior in comparison to other devices when sampling for VOCs (lmbrigiotta et al., 1988). Therefore, syringe samplers are not recommended.

3.2.2 Bladder Pumps
A bladder pump consists of a flexible bladder inside a rigid housing. Water enters the bladder from the bottom and is squeezed to the surface through a discharge line by gas pressure applied to the outside of the bladder. An air compressor and regulator turn the pressure on and off, allowing new water to enter the bladder and the cycle is repeated. The separate bladder chamber does not allow the sample to come in contact with the compressed air. Check valves at the top and bottom prevent backwash from the sample tube and bladder. Flow can be readily controlled and low rates of 100 millimeters per minute (ml/min) are easily obtainable. Teflon® bladders and Teflon®/stainless steel outer shells are recommended.

Bladder pumps have been used to depths greater than 200 feet and are available in sizes designed for 2-inch or smaller wells. The need for a power source and compressed air limits their mobility. Portable power sources are commercially available.

Potential problems with bladder pumps include sediment damaging the inner bladder and high suspended solids concentrations causing failure of check valves for some models (Nielsen and Nielsen 2006). Strainers or screens are available that attach below the bladder to filter material. Note that samples collected through a strainer or screens are not considered to be filtered.

Bladder pumps are generally recognized as the best overall sampling device for both inorganic and organic constituents (U.S. EPA, 1992). Muska et al., (1986) found that bladder pumps generate reproducible analytical results. Kasper and Serkowski (1988) concluded that the sampling rate and reliability of the bladder pump outperformed both the gas and mechanically driven piston pumps. Tai et al. (1991) concluded that a bladder pump yielded representative recoveries of VOCs compared to a control sample. Pohlmann and Hess (1988) determined that bladder pumps are suitable for collecting samples for almost any constituent.

Bladder pumps are recommended for purging and sampling. Whenever possible, the pump should be dedicated to the well. Doing so eliminates the need to transport and decontaminate the pump, thereby reducing the potential for cross contamination as well as saving time and reducing project cost.

3.2.3 Electrical Submersible Pumps
A variety of electrical submersible pumps are available. In the past, electrical submersible pumps were primarily designed for use in water supply wells and were difficult to use for contaminant monitoring purposes. However, manufacturers have since designed low-flow electrical submersible pumps for 2-inch diameter monitoring wells that can collect representative samples. Submersible pumps designed for ground water sampling incorporate non-sorptive materials (e.g., stainless steel, Teflon®, etc.) that are appropriate for collecting VOCs and other sensitive parameters. One disadvantage is that the heat generated by the motor could increase sample temperature, resulting in the loss of dissolved gases and VOCs and subsequent precipitation of trace metals (Nielsen and Nielsen, 2006) Therefore, after sampling, it is recommended that a sample be withdrawn and the temperature measured to assess whether the pump has increased the water temperature. Another disadvantage is the number of intricate parts, which may cause decontamination and maintenance to be time-consuming and difficult.

Two types of submersible pumps available are the centrifugal and the progressive cavity (helical-rotor) pumps. Both are positive displacement devices.

3.2.3.1 Centrifugal Submersible Pump
Centrifugal submersible pumps designed for 2-inch monitoring wells are usually cooled and lubricated with water rather than hydrocarbon-based coolants and lubricants that could contaminate samples. The electric motor spins or rotates an impeller (or series of impellers) that causes water to be accelerated outward and then

upward into and through the pump's discharge lines. The higher the pumping rate, the greater the potential for sample alteration by agitation, increased turbulence, and pressure changes. Therefore, a variable-speed centrifugal submersible pump capable of low-flow purging and sampling is essential for collecting a representative sample. Low-flow centrifugal submersible pumps appear to perform similarly to low-flow bladder pumps with respect to preserving sample integrity.

3.2.3.2 Progressive Cavity (Helical-Rotor) Pumps

Progressive cavity (helical-rotor) pumps are appropriate for collecting sensitive samples if low-flow pumping rates are used. An electric motor at the base turns a corkscrew-like helical rotor near the top. The helical rotor causes an upward movement of water trapped in the vacuities of the rotor and the water moves up and through the discharge line. A check valve at the top ensures that water in the discharge line (sampling tube) does not re-enter the pump. A controller box at the surface allows for variable flow rates.

3.2.4 Gas-Driven Piston Pumps

Although not commonly used, the gas-driven piston pump is acceptable if the parts contacting samples are chemically inert (i.e., will not affect sample representativeness). This device utilizes gas pressure to drive a piston between two chambers, one for gas and one for water. Gas is injected through one of two tubes to lower the piston in the gas chamber, allowing water to fill the upper water chamber. Pressure is then applied to a separate tube that pushes the piston upward and propels the sample to the surface. Water and gas remain separated. These pumps can operate at great depths and collect large-volume samples. Disadvantages are that valves and pistons are known to be damaged by fine-grained sediments and mobility is limited by the need for a gas supply. Additionally, the valving mechanism may cause a series of pressure drops that could cause sample degassing and pH changes (U.S. EPA, 1992).

3.2.5 Suction Lift Pumps

Suction lift pumps deliver samples by applying a vacuum at the surface. The negative pressure is applied by a portable pump attached to a tube lowered into the well. Suction pumps are limited by practical suction limits, which restrict their use to wells with water to approximately 25 feet below ground.

Surface centrifugal and *peristaltic* are the two major types of suction lift pumps. The peristaltic offers greater advantages over the surface centrifugal. Surface centrifugal pumps must be primed before being operated and should employ a vacuum flask to prevent contact of the sample with moving parts. Peristaltic pumps are self-priming and create a vacuum by a series of rotating wheels that compress the sample tubing. As the sample only contacts the tubing when using a peristaltic pump, no moving parts need to be decontaminated. Usually, disposable tubing is used. Peristaltic pumps generally cause less agitation than surface centrifugal pumps.

Suction lift pumps are very portable, widely available, and relatively inexpensive. Flow rates are controlled easily, providing adequate rates for sampling. These devices typically can be used in wells of any diameter and plumbness. Concerns with use of peristaltic pumps for sampling VOCs include potential for degassing due to negative pressures, potential for sorption of contaminants to flexible tubing, and potential for contribution of organic compounds from the flexible tubing (Striggow et al., 2017). The National Council of Industry for Air and Stream Improvement (NCASI, 1984) found a 10 to 30 percent loss in VOC concentrations from peristaltic/vacuum flask systems compared to results for bailers, bladder pumps, or submersible pumps. Imbrigiotta et al. (1988) also attributed losses of VOCs due to the vacuum created by peristaltic pumps.

To address these concerns, Ohio EPA recommends:

1. Use Compatible Tubing

Using compatible tubing reduces the risk of losses of VOCs due to adsorption to the tubing or addition of contaminants from the tubing (i.e., material leaching).

- Ensure the sample tubing (drop-tubing / discharge tubing) is compatible with COCs being sampled.
  - Low-density polyethylene (LDPE), high-density polyethylene (HDPE), and Teflon®-lined tubing are all commonly used and generally compatible with most COCs (VOCs, semivolitile organic compounds [SVOCs], metals). *NOTE: Teflon® should not be used to sample for PFAS.*
- Ensure the flexible tubing (pump head tubing) is compatible with COCs being sampled.
  - The MasterFlex® flexible tubing typically used is compatible with a wide range of COCs (VOCs, SVOCs, metals, and can be compatible with PFAS)

2. Use New or Dedicated Tubing at Each Sample Location

Use of new or dedicated tubing at each sampling location prevents cross-contamination that could occur from reused tubing.

3. Look for off-gassing of sample (bubbles in the sample tubing).

Look for significant bubbles forming in the sample tubing to determine whether off-gassing is occurring due to the use of the peristaltic pump. Off-gassing (methane, CO2, etc.) can cause VOCs to move out of solution and into the entrained bubbles, causing unrepresentative low results. Dissolved VOCs exposed to lower atmospheric pressure of peristaltic pumps can also degas, biasing results low.

Off-gassing of CO2 or other dissolved gasses can alter the geochemical conditions (such as pH) of the water in the tubing, potentially chemically altering metals or other redox-sensitive parameters in the sample.

- If bubble formation in tubing is significant, either:
    - stop sampling with peristaltic and return with alternative method (i.e., bladder pump)
    - if first-time sampling event with additional sampling expected, consider collecting the samples but qualifying the data "J" as estimated low and sample with an alternative method (i.e., bladder pump) during follow-up or confirmation sampling events.
- If no significant bubble formation is observed in the sample tubing, use of the peristaltic pumps is OK.

4. Incompatible flexible tubing considerations. (Rare situation)

If there is a known compatibility issue identified with the flexible tubing, but the sample tubing is compatible, the following alternative sampling methods could be employed to prevent contact with the flexible tubing (i.e., sample collected before the pump head).

- VOCs could be sampled using the "soda straw" method. *NOTE: When purging low-flow and sampling using the soda straw method, if additional sample volume is needed and the tubing must be replaced in the well, stabilization of that well must start over due to the disturbance of the standing water column in the casing.*

- SVOCs and some metals (e.g., non-VOC samples collected in 1-Liter amber glass jars) could be collected using the "vacuum transfer cap" method.

Peristaltic pumps may not be a preferred sampling method if sufficient dissolved CO_2, dissolved methane, or other dissolved gasses are present in ground water being sampled. Negative pressures created by the pump may cause excessive degassing of dissolved gasses. Degassing of dissolved CO_2 during purging can cause pH changes that may affect dissolved metals concentrations in samples and degassing can create space for volatilization of target VOCs into the entrained gases. If gas bubbles are observed in sample tubing during stabilized pumping rates, then significant degassing may be occurring that could affect sample quality.

If samples are being collected for analyses for low concentrations of organics and there is concern over the potential interference due to contact with the flexible tubing, then they should be collected in-line, ahead of the pump, and a sufficient volume of water should be pumped through the system to account for the initial filling of the containers when a negative head space was present. US EPA provides the following two methods to sample water before the pump-head flexible tubing to collect samples for organics analyses using a peristaltic pump (Striggow et al., 2017):

"Soda straw" method:	Once the well is sufficiently purged, the sample tubing is crimped (typically in a double-z configuration) to prevent the water in the sample tubing from flowing back into the well. The sample tubing is withdrawn from the well and sample containers are filled from the water stored within the tubing by carefully releasing the crimp to allow water to flow. U.S. EPA provides a complete procedure for the soda straw method in Striggow et al., 2017).
Vacuum transfer cap method:	A vacuum transfer cap is added to the sample tubing before the flexible tubing to allow for collection of a sample prior to contact with the flexible tubing. Several vacuum transfer cap devices are commercially available. Due to the potential for degassing of VOCs from samples, this method is only recommended for sampling SVOCs or other sample analyses requiring 1-liter glass containers.

Samples collected for metals and inorganic analyses using peristaltic pumps may be collected through the flexible tubing, provided that a rinse blank of the flexible tubing is also collected. Substantial degassing of CO2 can result in pH shifts that can affect the quality of samples collected for metals and inorganic analyses. (Striggow et al., 2017).

3.2.6 Passive Diffusion Samplers

Passive diffusion bag samplers (PDBs) use a LDPE diffusion membrane filled with deionized water to collect water samples for VOC analysis. The polyethylene acts as a semi-permeable membrane allowing volatile contaminants to diffuse into the deionized water. Once chemical equilibrium is reached, a water sample that is representative of the VOC concentrations may be obtained from the interval at which the sampler is placed.). Use of multiple PDB samplers at different depths within a well screen interval can allow for a vertical profile of the VOC contamination within the well.

Equilibration time for diffusion samplers depends on the time required for the environment disturbed by deployment of the sampler to return to ambient conditions and time required by the sampler to equilibrate with the ambient water. To account for this, PDB samplers are generally deployed a minimum of 14 days prior to sample collection. Some lower conductivity formations may require more than two weeks (Striggow et al., 2017; Vroblesky and Campbell, 2001). Samplers can be left in from one sampling event to another then removed and replaced with a new sampler to minimize mobilization and maximize efficiency. Although the sampler may be deployed for an extended period (e.g., three months or longer); the analytical results will be the average of the sample equilibration time for the analyte of concern (generally the last 1 to 4 days).

Advantages of PDB sampling include its low cost, minimal purging and water disposal, and the ability to monitor a variety of VOCs. A disadvantage is that they are not applicable to inorganics and other contaminants that do not readily diffuse across the semi-permeable membrane. PDB sampling may not be applicable for sites where water in the well casing may not be representative of the saturated zone adjacent to the well screen. This may occur when water in the well casing is stagnant, or when there is a vertical flow within the well. In addition, PDB samplers do not provide a discrete time-interval sample, but rather an average of the concentrations in the well over the equilibrium period.

Passive diffusion bag samplers are appropriate for long-term monitoring at well-characterized sites. The target analytes should be limited to chemicals that have been demonstrated to diffuse well through polyethylene (i.e., most VOCs and limited non-VOCs), as listed in Tables 1-1 and 4-1 of ITRC's PDB sampler guidance document (ITRC, 2004). A combined version of these tables is provided below as a reference (Table 10.2). However, as the compound list may change as further tests are conducted, ITRC *http://www.itrcweb.org/* should be referenced for the most recent list of chemicals favorable for sampling with PDB. The site sampled should have sufficient ground water flow to provide equilibrium between the water in the well screen and the surrounding ground water zone. ITRC (2004) suggests that care should be given in interpreting PDB results when the hydraulic conductivity is $<10^{-5}$ cm/s, the hydraulic gradient is <0.001, or the ground water velocity is < 0.5 ft/day. Use of PDBs is not appropriate when a vertical flow in the well exists. A deployment time of at least two weeks is recommended to allow for diffusion of the analytes across the membrane (ITRC, 2004, Vroblesky, 2001; Vroblesky and Hyde, 1997; Yeskis and Zavala, 2002; and USGS, 2014).

3.2.7 Equilibrated Grab Samplers

Equilibrated Grab Samplers are devices that can be used when no purge techniques are acceptable (Section 4.5.4 – Minimum/No Purge Sampling). They are placed in the well before sampling and remain closed. The water is collected when the sampler is activated. Samples are either transferred to containers at the well head or the sampler is shipped to the laboratory for analysis. Examples of equilibrated grab samples include Hydrasleeve[TM], Snap Sampler[TM], and Kemmerer Sampler (USGS, 2014; Striggow et al. 2017).

Table 10.2 Compounds tested with PDB in laboratory and field tests.

Favorable laboratory diffusion testing results		
Benzene	1,3-Dichlorobenzene	Napthalene*
Bromodichloromethane*	1,4-Dichlorobenzene	1,1,2,2-Tetrachloroethane
Bromobenzene**	Dichlorodifluoromethane	Tetrachloroethene (PCE)
Bromochloromethane**	1,1-Dichchloroethane**	Toluene
Bromoform*	1,2-Dichloroethane	1,2,3-Thrichlorobenzene**
n-Butylbenzene**	1,1-Dichloroethene	1,2,4-Trichlorobenzene**
sec-Butylbenzene**	cis-1,2-Dichloroethene	1,1,1-Trichloroethane
tert-Butylbenzene**	trans-1,2-Dichloroethene	1,1,2-Trichloroethane
Carbon disulfide**	1,2-Dichloropropane	Tricholoroethene (TCE)
Carbon tetrachloride	cis-Dichloropropene*	Trichlorofluoromethane
Chlorobenzene	1,2-Dibromoethane*	1,1,2-Trhichloro-1,2,2-
Choroethane	trans-1,3-Dichloropropene*	trifluoroethane**
Chloroform*	Ethyl benzene	1,2,3-Trichloropropane*
Chloromethane	Hexachlorobutadiene**	Vinyl chloride
2-Chlorovinylether*	p-Isopropyltoluene**	m,p-Xylene**
Dibromochloromethane	1-Methylethylbenzene**	o-Xylene**
Dibromomethane		Xylenes (total)
1,2-Dichlorobenzene		
Unfavorable diffusion testing results		
Acetone*	Methyl tert-butyl ether*	Styrene*
tert-Amyl methyl ether**#	Naphthalene**	1,2,4-Trimethylbenzene**
Bromoform**#	n-Propylbenzene**	1,3,5-Trimethylbenzene**
Methyl iso-butyl ketone*		

 * Laboratory results only, (Vroblesky 2001a)

 **Results from field tests only, (Parsons 2003)

 #The data set for this compound was relatively small (fewer than five instances of comparison), so the power of the classification (i.e.,
   acceptable, or unacceptable) is fairly low.

3.2.8 Other Devices

The *gas drive sampler* operates by applying positive gas pressure into a sample chamber to force the water to the ground surface. Water enters through a valve at the bottom of the sampler into the sample chamber. When pressure is applied, the valve closes, and water is forced through a discharge line to the surface. When the pressure is reduced, the valve reopens, allowing water to enter the chamber, and the cycle is repeated.

Gas drive samplers are available with inert components and in a variety of diameters. They can provide continuous flow at acceptable rates for sampling. Their major drawback is that they allow for gas contact with the sample, which can cause the loss of dissolved gasses and VOCs and potentially other chemical alterations. Gas can also mix with the sample, causing further alteration. For these reasons, use of these samplers is generally not recommended. Additionally, mobility is limited by the need to provide compressed gas. When sampling very deep wells, high gas pressures are needed, and the device should be designed to handle this added stress.

Gas lift samplers inject air or gas into the water column to "blow" water to the surface. The gas acts as a carrier fluid; however, the gas (even if inert) causes degassing and volatilization. Additionally, aeration and turbulence can further alter the original water chemistry (Lee and Jones, 1983). Therefore, gas lift systems are unacceptable.

The *inertial lift pump* consists of a foot valve at the end of a flexible tube that runs to the surface. At the beginning of sampling, the water column in the sampling tube is equal to that in the well. A levered handle, a gasoline motor drive, or manual operation provides a continuous up-and-down movement of the tubing. An initial rapid upstroke lifts the water column in the tubing a distance equal to the stroke length. At the end of the upstroke, the water continues to move slightly upward by inertia. On the down stroke, the foot valve opens allowing fresh water to enter the tube. Inertial lift pumps can be used for both purging and sampling; however,

these devices cause a surging action that may cause increased turbidity, loss of volatiles, aeration, and degassing of samples. These pumps can be used when collecting non-sensitive samples; however, should not be used for VOCs or gas-sensitive samples (Striggow et al., 2017). Table 10.3 summarizes the recommended sampling mechanisms for the targeted analytes.

3.2.9 Use of Packers

Packers are inflatable rubber devices used in a well or open borehole to isolate water-bearing intervals for hydraulic testing or ground water sampling. Packers can be used to minimize purge volumes in wells with long intake columns by isolating the sampled zone from stagnant water above the screen. Both single and double packer assemblies are used. For sampling, a pump is typically installed above or below a single packer or within a double packer assembly with a discharge line extending through the upper packer. Packer assemblies may include a drop tube through which water level tapes, transducers, pump control and discharge lines, and other monitoring and sampling equipment may extend to the isolated interval. Prior to using packer assemblies for sampling, all potential limitations or problems should be carefully evaluated and resolved, and the use of packers should be justified. For example, packer materials selected should not leach or sorb contaminants. In addition, the water level within the packer interval should not be drawn down below the upper packer. The potential for vertical movement of ground water to or from the packer interval outside of the well or borehole should be evaluated, as well as the potential for leakage around the inflated packers. For additional discussion on packers, refer to Oliveros et al. (1988).

3.2.10 Equipment Limitations and Considerations for sampling PFAS

The widespread use of PFAS in materials and manufacturing of ground water sampling equipment creates the potential for cross-contamination that should be considered and minimized to the extent possible. Typical analyses for PFAS are conducted to detect part-per-trillion concentrations, which makes any cross-contamination potential a critical quality assurance issue.

Sampling equipment materials having direct contact with water samples should be both inert and free of PFAS. These can include materials used in sample tubing, pump seals (i.e., o-rings or gaskets), plastic bladders, etc. (KEMI, 2015).

Consideration should also be taken for materials, such as personal protective equipment (PPE), that will not come into direct contact with water samples, but which could create cross-contamination potential. Certain materials that should be avoided may include waterproof textiles (i.e., GoreTex® raingear, sunscreens (KEMI, 2015; Fuji, 2013).

Only certified PFAS-free water supplied by the laboratory should be used for preparation of field reagent blanks and equipment blanks (NGWA, 2018). Certified PFAS-free water may also be needed for equipment decontamination unless any alternative water source used has been adequately tested.

To prevent cross-contamination, PFAS and fluoropolymer-free equipment should be used. If the composition of materials used in sampling equipment is not known, the manufacturer of the equipment should be contacted to determine equipment compatibility with PFAS sampling. The following list of the materials that could be commonly encountered in films, coatings, seals, tube linings, etc. that should not be used when sampling for PFAS (KEMI, 2015) includes, but is not limited to:

- Polytetrafluoroethylene (PTFE / Teflon®)
- Perfluoroalkoxy polymer (PFA)
- Polyvinylidene (PVDF)
- Polychlorotrifluoroethylene (PCTFE)
- Ethylene-tetrafluoroethylene (ETFE)
- Fluoroelastomers such as Viton®
- Fluorinated ethylene propylene (FEP)

As an "emerging contaminant", guidance, recommendations, and operating procedures for sampling may change as more PFAS sampling information and experience is gained. When preparing for PFAS sampling, Ohio EPA recommends consulting the most recent U.S. EPA and Ohio EPA PFAS sampling guidance documents.

Table 10.3 Recommended Ground Water Sampling Techniques for Compliance or Risk Assessment Samples

SAMPLING MECHANISM	Target Analytes			CONSIDERATIONS FOR USE
	VOC	SVOC	INORGANIC	
Bladder Pump	X	X	X	overall, most suitable mechanism for most analytes (Pohlmann and Hess, 1988)
Electric Submersible Pumps	X	X	X	potential for alteration by elevated temperature as pump heats up (Imbrigiotta et al., 1988)
Suction Lift Pump (Peristaltic / Centrifugal)	X	X	X	potential for degassing due to negative pressure; must ensure compatible tubing materials; special sampling methods may be needed to prevent interferences (Imbrigiotta et al., 1988; Striggow et al., 2017)
Bailers	X	X	X	potential agitation, degassing, increased turbidity, and surging may affect sample quality and well development; carefully controlled purging and sampling may yield acceptable results (Imbrigiotta et al., 1988; Striggow et al., 2017)
Gas Driven Piston Pumps	X	X	X	may be acceptable if gas is non-contact with sample; valving mechanism may cause pressure drops and degassing (U.S. EPA, 1992)
Syringe Samplers		X	X	potential for degassing; not used for purging; limited sample volume (Imbrigiotta et al., 1988)
Inertial Lift Pumps		X[1]		potential agitation, degassing, increased turbidity, and surging may affect well development; not recommended for VOCs or gas-sensitive samples (Striggow et al., 2017)
Passive Diffusion Bag Sampler	X[1]	X[1]		limited list of acceptable target analytes; may be appropriate for long-term monitoring at well-characterized sites where no-purge is determined to be an acceptable approach (Striggow et al., 2017)
Equilibrated Grab Sampler	X	X	X	option when no-purge is determined to be an acceptable approach at a site (Striggow et al., 2017)

NOTE:

X = may be suitable for listed target analytes
Gray Shaded = not suitable for target analytes
[1] = may not be suitable for all analytes in this category
Table does not address gas lift pumps. These devices are generally not recommended.

4.0 Field Procedures

Prior to the sampling event, all applicable site-specific plans, such as health and safety plans and sampling and analysis plans should be reviewed by the sampling crew. The objective of the sampling event is to collect representative ground water samples. Therefore, the applicable procedures below should be completed and documented to ensure that representative ground water samples can be obtained. (Documentation guidance is provided in subsequent parts of this chapter).

4.1 Field Conditions

Weather and site-specific conditions that could affect sample representativeness should be documented. The approximate ambient air temperature, precipitation, and wind and other field conditions should be noted in a field notebook or field sampling form. In addition, any site-specific conditions or situations that could potentially alter the ground water samples or water level measurements should be recorded. Examples include but should not be limited to excavation or construction activities, accidental spills, and presence of smoke, vapors, or air contaminants from anthropogenic activities.

4.1.1 Operator actions and attire

Activities that are necessary for the sampling crew that may affect the sampling results, such as fueling a vehicle or generator, should be performed taking care that the risk of sample contamination is minimized, such as refueling away from the sample site and re-gloving with new gloves and washing hands after using fuels.

Sampling personnel should not use perfume, hand lotion etc. prior to collecting a ground water sample. If insect repellent is necessary, the repellent should be applied at least one hour before sample collection and care should be taken not to allow the repellent to come into contact with the sampling equipment and it should be recorded that insect repellent was used (Wilson, 1995). Be aware that some personal hygiene products should be avoided when sampling for PFAS compounds (Section 3.2.10). If conducting trace metal analyses, all metal in proximity to sampling should be limited, including apparel.

4.2 Well Inspection and Preparation

Upon arrival, the well protective casing, cap, and lock should be carefully inspected, and observations recorded to document whether damage or tampering has occurred.[5] Cracks in the casing and/or surface concrete seal should be noted, as well as soil washouts and depressions around the casing.

Before taking any measurements, weeds and/or debris that may interfere with the sampling should be cleared from the well area. Equipment that could come into contact with the ground water or contaminated soil should be covered and stored off the ground to avoid potential cross-contamination. If clean plastic sheeting is placed on the ground to help prevent contamination of equipment then care should be taken to prevent slips, trips or falls. Any plastic sheeting should be disposed properly following completion of sampling at each well. A portable field table covered with a new plastic sheet at each well is convenient for preparing equipment and performing field measurements (Wilson, 1995).

4.3 Well Measurements

Appropriate measurements should be made before any water is purged and sampled. These include measuring of static water levels and total well depth, and depending on site-specific conditions or circumstances, detection of gases, organic vapors and immiscible liquids.

4.3.1 Detection of Organic Vapors and Gases

Because VOCs often present health and safety concerns, field screening instruments are often used for field monitoring if ground water is contaminated with VOCs. Field screening instruments include photoionization detectors (PIDs) and flame-ionization detectors.[6] PIDs and FIDs are typically used to provide an estimate of the total volatile organic vapor concentration (e.g., benzene, vinyl chloride, tetrachloroethane), rather than a

---

[5] See Chapter 8 (Well Development, Maintenance, and Redevelopment) for additional information on periodic well maintenance checks and well-integrity tests).

[6] For further information on types and uses of these instruments, see Anastas and Belknap (1980), Brown et al. (1980) and DuBose et al. (1981).

quantitative result for individual compounds. FIDs are capable of detecting methane, while PIDs are not. The selection of the correct lamp is important when using a PID meter. Vapor measurements can give useful information about potential ground water quality and allow for sampling personnel to take appropriate safety precautions. It also may be useful to determine the potential for the presence of immiscible layers, which necessitate additional sampling procedures and concerns.

Gases that typically may be of concern include methane, CO_2, and H_2S. Field meters are available for detecting methane, CO_2, and hydrogen sulfide, including combination meters that can be used to screen for two or more of these gases. Generally, methane and CO_2 may occur in monitoring wells at solid waste landfill facilities. Methane may also be present as natural gas in bedrock formations. The presence of methane is significant because it may include trace amounts of VOCs that are too low to be detected with a PID or FID. Additionally, methane is a health and safety concern because it can cause a potentially explosive atmosphere. CO_2 may affect ground water chemistry by altering pH or alkalinity. H_2S, which is typically associated with sewage or decaying vegetation, may affect pH meter performance. H_2S gas can also be naturally occurring in carbonate bedrock aquifers. H_2S presents health and safety concerns at concentrations as low as 1 ppm in air.

4.3.2 Water Level

In addition to providing hydrogeologic information on a continuing basis, measurement of the water level in a well enables determination of the volume of water contained, which may be useful for purging determinations. Measurements should be taken from the entire well network before any water removal to obtain a single "snapshot" of current hydraulic head conditions and to avoid potential effects on the water levels in nearby wells. The measurements should be made within a period of time short enough to avoid temporal variations in ground water flow that could preclude an accurate determination of ground water flow rate and direction. The period of time should not exceed 24 hours.

Measurements can be taken manually or automatically. Table 10.4 summarizes the manual methods. Automatic, continuous recording devices may be useful for collection of long-term data and in pumping tests. Water level measurements are described in more detail by Dalton et al. (1991), Aller et al. (1991), and ASTM D4750. An electronic probe is recommended for taking water level measurements.

Measurements should generally be within ±0.01 ft. There may be instances where this level of accuracy is not necessary (e.g., steep water table, wells are far apart); however, accuracy may be required by rule. All wells should have accurate surveyed reference points[7] for water level determination. Typically, a marked point on the top of the inner riser pipe is used.

Equipment should be properly decontaminated before use in each well to ensure sample integrity and prevent cross-contamination. Techniques are discussed later in this chapter.

LNAPL (see below) may affect the water level measurements in a well. It is important to know the density of the free product because water level measurements in monitoring wells that also contain free product should be corrected to account for the different densities of water and product and the thickness of the product layer. See U.S. EPA (1996a) for procedure to correct for an LNAPL layer.

---

[7]Ohio EPA recommends that the reference point be based on the National Geodetic Vertical Datum or local common datum. However, an arbitrary datum common to all wells in the monitoring network may be acceptable if necessary.

4.3.3 Well Depth

Measuring the depth of a well indicates the amount of siltation that has occurred. Natural siltation can block water from entering, which could lead to erroneous water level measurements and bias analytical results by increasing sample turbidity. Checking depth also provides a check on casing integrity.

Depth can be determined with a weighted tape measure or marked cable, each of which should be composed of inert materials. Often, the same device that is used to measure water levels can be used. Heavier weights are necessary as depth increases to effectively "feel" the well bottom. The measurement should be recorded on the field log.

Measuring the well depth every time water levels or samples are obtained is generally not necessary. Obtaining a depth from a well with a dedicated pump may not be possible (and is not advisable) unless the pump is removed. In addition, the logistics of decontaminating the entire length of the measuring tape in contact with contaminated ground water may cause depth measurements to be impractical. At minimum, depth measurements should be taken once a year in wells that do not have dedicated pumps. Measurements in wells with pumps should be taken whenever the pump is removed for maintenance. If siltation is suspected to be a problem (e.g., noted increase in sample turbidity, or decrease in pump efficiency), the pump should be removed, and the well depth checked. Depth measurements should be to the nearest 0.1 foot (U.S. EPA, 2001a). Depth to bottom can be obtained when collecting the round of depth-to-water measurements. Care should be taken to avoid stirring up any accumulated sediments, thus increasing turbidity of the water column. Additional methods for measuring water levels were documented in previous versions of this table but have been removed as the methods are no longer commonly used. The removed methods include weighted steel tape with chalk, airline, float, popper acoustic probe, radar and laser.

Table 10.4 Summary of methods for manual measurement of water levels (based on Dalton et al., 1991, ASTM D4750 and U.S. EPA, 2001a).

MEASUREMENT METHOD	MEASUREMENT ACCURACY (in feet)	DESCRIPTION & ADVANTAGES	MAJOR INTERFERENCES OR DISADVANTAGES
NON-FLOWING WELLS			
Electrical method	0.01 to 0.1	An electronic probe is lowered into the well. When the probe comes into contact with water, the electric circuit in the probe is completed and an audible and/or visible signal is given.	• Errors result from changes in cable length as a function of use, temperature and depth • Reliable contact may be difficult if LNAPLs are present or water has low conductivity
Transducer	0.01 to 0.1	A transducer is lowered a known distance into the well and allowed to equilibrate with fluid temperature. Distance of submergence of the transducer is read on the signal conditioning unit and is subtracted from the cable length referenced at the top of the well.	• Accuracy is dependent upon range and sensitivity of the device.
Ultrasonic	0.02 to 0.1	Water level measurements are determined by an instrument that measures the arrival time of a reflected transmitted sonic or ultrasonic wave pulse.	• Accuracy can be limited by the change of temperature in the path of the sound wave and other reflective surfaces in the well (i.e., casing, pumps, etc.). Greater depth, the less accurate.
FLOWING WELLS			
Casing Extension	0.1	A simple extension is attached to the well casing to allow water level to be measured directly.	• The device is only practical when additional height requirement is only several feet. • Accuracy low because water level in flowing wells tends to fluctuate.
Manometer/ Pressure Gauge	0.1 to 0.5	The pressure of water within a sealed or "shut-in" well is measured.	• Gauge inaccuracies. • Calibration is required.
Pressure Transducers	0.02	Procedures are the same as described above for transducers. The range of a pressure transducer should be carefully matched with shut-in well pressure.	• Changes in temperature in the transducers cause errors.

4.3.4 Detection of Immiscible Liquids

NAPLs are organic liquids that exist as a separate, immiscible phase when in contact with water and/or air. If the presence of NAPLs is suspected, the sampling program should include devices and protocols to detect them. DNAPLs are referred to as "sinkers" because their density (greater than water) causes them to sink. LNAPLs are referred to as "floaters" because their density (less than water) causes them to float on the water table surface. If floaters are of concern, it is important that, upon opening the well cap, the air in the casing is monitored with a PID or an OVA. In addition to providing information on worker health risks, air monitoring can serve as a first indication of the presence of volatile floaters.

Protocol to detect immiscible liquids should always include visual inspection of purged water and any equipment that is removed from the well after use. Additionally, probes and reactive pastes have been developed to determine air/immiscible and water/immiscible interfaces. Indicator pastes are used to coat an interface probe or a weighted tape. An observed reaction indicates the presence of an immiscible liquid. Probes and pastes can be utilized for detecting both floaters and sinkers (U.S. EPA, 1992). Transparent bailers also can be used.

4.4 Sampling Immiscible Liquids

If an LNAPL is found to be present, a bailer or submersible pump can be used to remove it, if necessary (U.S. EPA, 1992). Any LNAPL greater than 2 feet in thickness can be evacuated using a bottom-valved bailer. The bailer should be lowered slowly to a depth less than the product/water interface. A modified, top-filling bailer (bottom valve sealed off with a fluorocarbon resin sheet between the ball and ball seat) can be used to remove immiscible layers less than 2 feet in thickness.   In either case, a peristaltic pump also can be utilized if depth to product is less than 25 feet. Any LNAPL less than 2 inches thick can be collected from the top of the water column using a bailer (U.S. EPA, 1992). Samples collected in this manner consist of both an aqueous and non-aqueous phase.

To the extent possible, the sampling and purging method should prevent the disturbance of DNAPL. A sample of the DNAPL should be obtained after the ground water sample has been obtained. Double -check valve bailers, Kemmerer devices and syringe samplers often are used.

When an immiscible layer is to be analyzed, additional sampling equipment (i.e., containers) may be needed to have sufficient volume for laboratory analysis. It is important that appropriate QA/QC procedures be followed when collecting samples of any immiscible liquids. If any immiscible layer is removed, it should be properly collected, containerized, characterized, and managed. The Ohio EPA Division of Environmental Response and Revitalization (DERR) Resource Conservation and Recovery Act (RCRA) Program can be contacted for guidance on these issues.

4.5 Purging & Sampling Procedures

Upon completion of the preliminary procedures, purging and sampling of ground water can generally be accomplished by volumetric or low flow rate methods. However, volumetric purging and low flow rate purging/sampling may not be feasible for wells that produce less than 100 ml/min. Therefore, other approaches should be considered, including minimum/no purge sampling as well as purging to dryness and sampling as soon as the well has recharged sufficiently. These approaches are discussed below, along with methods to determine when purging is complete by measuring indicator parameters.

Where dedicated equipment is not used, sampling should progress from wells least likely to be contaminated to those most likely to be contaminated to minimize the potential for cross-contamination. Decontamination of down-well equipment should be considered as discussed in Section 4.9 below. Care needs to be taken to avoid agitation and temperature increases during sample collection.

4.5.1 Field Measurements of Ground Water Indicator Parameters

Indicator parameters are measured in the field to evaluate well stabilization during purging, provide information on general ground water quality, help evaluate well construction, or indicate when well maintenance is needed. Indicator parameter data also may be helpful in evaluating the presence of ground water contamination. Regulated entities (such as municipal or hazardous waste landfills) may require water quality indicator parameters as part of a sampling program. However, measurement of field parameters is not always required or possible. Indicator parameters to evaluate well stabilization during purging may not be necessary when recovery is slow and the only viable sampling methods are purging to dryness or passive techniques.

Indicator parameters include specific conductance, pH, DO, ORP, temperature, and turbidity. Due to their unstable nature, field measurements of indicator parameters are preferred over laboratory analyses, which may be less representative of in-situ ground water conditions.

Specific conductance measures the ability of water to conduct an electric current. For most circumstances, specific conductance has been demonstrated to be a reliable indicator of the chemical stabilization of purge water (e.g., Barcelona et. al., 1994).[8] For ground water, it is generally reported in micromhos per centimeter (µmhos/cm), as natural waters commonly exhibit specific conductance well above 1 µmhos/cm (Hem, 1992). Specific conductance is a relative measure of the number of ions present in ground water, as the magnitude of the current conducted by a ground water sample is directly proportional to its ionic concentration. Based on this relationship, total dissolved solid concentrations may be approximated from specific conductance data (Hem, 1992). High readings may indicate contamination, especially if the readings are elevated compared to background. Alternatively, elevated specific conductance may indicate grout contamination in a well or an inadequate grout seal that is allowing infiltration of surface water or ground water from overlying saturated zones. Elevated specific conductance readings may also indicate inadequate well development (Garner, 1988).

pH is a measure of the effective concentration (or activity) of hydrogen ions and is expressed as the negative base-10 logarithm of the hydrogen-ion activity in moles per liter (more commonly reported as S.U.). Uncontaminated ground water typically exhibits a pH ranging from 5 S.U. to 9 S.U. (Brownlow, 1979; Ohio EPA, 2003). While pH has commonly been used as a purge water stabilization indicator, it is less sensitive than specific conductance or oxidation-reduction potential in distinguishing stagnant casing water from formation water. However, pH measurements are important for the interpretation of ground water quality data (Puls and Barcelona, 1996), as pH indicates the relative solubility of metals and speciation of many other chemicals (Garner, 1988). First, pH measurements reflect chemical reactions that produce or consume hydrogen ions (Hem, 1992), and therefore, changes in pH from background may indicate the presence of ground water contamination or that existing contamination has spread. Second, pH is useful for identifying well construction or maintenance problems. For example, pH readings that consistently increase (7.8, 8.3, 8.8, 9.4...) during purging may indicate grout contamination in the sand pack and screened interval. pH is also highly dependent on local geology.

Dissolved oxygen **(DO)** has been demonstrated to be a reliable indicator of the chemical stabilization of purge water under most ground water purging and sampling circumstances (e.g., Barcelona et. al., 1994). DO is a good indicator when sampling for VOCs, because erratic or elevated DO readings may reflect procedures that are causing excessive agitation and aeration of the ground water being drawn from the well and subsequent loss of VOCs (Pennino, 1988). Artificially aerated ground water may also adversely affect dissolved metals analyses.

---

[8]Specific conductance should not be used by itself to determine whether adequate purging has been completed. Ohio EPA recommends using multiple indicator parameters to determine when to terminate purging and begin sampling regardless of the assumed reliability of the data.

Concentrations of DO in ground water (1 to 4 milligrams per liter [mg/l]) (Testa and Winegardner, 1991). Ohio EPA recommends measuring DO with a flow-through cell, although in some circumstances a hand-held meter may be acceptable. Natural ground water tends to have low DO conditions (e.g., < 1 mg/L) when there is not a hydraulic connection to the surface. Higher levels can be obtained when the pathways to the surface are short. Accordingly, relatively low DO concentrations (< 1 mg/l) in ground water may indicate the biodegradation of organic contaminants, including VOCs (U.S. EPA, 1997).

Oxidation-reduction potential (ORP), also referred to as redox potential, is a numerical index of the intensity of the oxidizing or reducing conditions within an aqueous solution such as ground water. ORP can be converted to Eh (Striggow et al., 2017; James et al., 2004). Oxidizing conditions are indicated by positive potentials and reducing conditions are indicated by negative potentials. ORP measurements are generally expressed in millivolts (mV). The ORP of natural (uncontaminated) ground water typically ranges from +500 to -100 mV (Brownlow, 1979). Ground water contaminated with organic compounds generally exhibits depressed ORP values compared to background conditions and may exhibit ORP values as low as -400 mV (Wiedemeier et. al., 1997). ORP may not be an appropriate stabilization parameter for some ground water conditions (Yeskis and Zavala, 2002). ORP data is useful for evaluating the expected oxidation state of metals and other chemical species in a general sense, especially when collected with pH data. This can also be used to determine expected solubility, based on oxidation state and specific chemical species present, by using Eh pH diagrams. Such information may be helpful for fate-and-transport modeling. However, aquifers and other saturated zones are open systems that are affected by many variables, and therefore, the actual chemical species present in ground water will not necessarily correspond to measured ORP and pH data (Hem, 1992; Rose and Long, 1988). In addition, ORP values cannot be used to derive or infer DO values, and vice versa (Rose and Long, 1988).

Temperature is not necessarily an indicator of ground water chemical stabilization and is generally not very sensitive in distinguishing between stagnant casing water and formation water (Puls and Barcelona, 1996). Nevertheless, temperature is important for data interpretation. Eh, pH, and other parameters are temperature dependent. For example, stabilized temperature readings that are representative of typical ground water conditions help demonstrate that the sample was collected in a manner that minimized exposure to elevated temperature variations, e.g., heating from the electric motor of a submersible pump. Elevating the temperature of a sample may result in loss of VOCs or the progression of chemical reactions that may alter the sample quality in an undesirable manner. Ground water temperatures in Ohio typically range from 9 to 13 degrees Celsius (°C) (Heath, 1987).

Turbidity, which is the visible presence of suspended mineral and organic particles in a ground water sample, can be useful to measure during purging. Relatively high or erratic measurements may indicate inadequate well construction, well development or improper sampling procedures, such as purging at an excessive rate that exceeds the well yield (Puls and Powell, 1992; Paul *et. al.*, 1988), or redox reactions occurring/changing oxidation state. Purging and sampling in a manner that produces low-turbidity water is particularly important when analyzing for total metals, which may exhibit artificially elevated concentrations in high-turbidity samples (Gibbons and Sara, 1993). Generally, the turbidity of *in-situ* ground water is very low (Nightingale and Bianchi, 1977). When sampling for contaminants or parameters that may be biased by turbidity, Ohio EPA recommends stabilizing the turbidity readings at or below 10 nephelometric turbidity units (NTUs) (Yeskis and Zavala, 2002). It is recognized that some ground water zones may have natural turbidity higher than 10 NTUs. If turbidity is being used as a stabilization parameter, it may be necessary to evaluate the stabilization criteria on a site-by-site basis. The recommended stabilization criteria for NTUs is 10 percent. In addition, turbidity can be used if high anomalous levels of Fe, Mn, or other metals are detected due to sample acidification and dissolution of solid constituents.

Table 10.5 provides stabilization criteria for each parameter discussed above. Ohio EPA recommends that specific conductance plus two additional parameters should be selected[9]. **A parameter can be considered stable when at least three consecutive readings have all stabilized within the stabilization criteria**

---

9 Regulatory program rules should be checked to determine whether the rules require specific indicator parameters.

range when compared to one another. For example, three pH readings of 6.80, 6.90 and 7.00 S.U. indicate stabilization because all three values are within +/-0.2 S.U. of one another. However, three pH measurements of 6.60, 6.80 and 7.00 S.U. do not indicate stabilization because two values (6.60 and 7.00) are greater than +/-0.2 S.U. of each other. The interval between measurements is discussed in the particular purging/sampling methodology section.

Field measurements performed to fulfill regulatory requirements, beyond those used to measure for stabilization, should be obtained after purging and before samples are collected for analysis. Portable field instruments should be used. Probes enabling down-hole measurement can be used and may increase data representativeness. All in-well instruments and probes should be appropriately decontaminated before use to prevent contamination of the well water. Flow-through cells can be used when sampling with pumps.

Table 10.5. Stabilization Criteria with References for Water-Quality Indicator Parameters (Yeskis and Zavala, 2002).

Parameter	Stabilization Criteria
pH	± 0.2 standard units
specific conductance	± 3%
oxidation-reduction potential (ORP)	± 20 millivolts
turbidity	less than or equal to 10 NTUs, or ± 10% if turbidity is > 10 NTUs
dissolved oxygen (DO)	± 10% of reading value or ± 0.2 mg/L, whichever is greater
temperature	± 0.5 º Celsius

Calibration of instruments should be performed in accordance with the manufacturer's instructions during each day of ground water sampling. A pH meter should be periodically calibrated with a two-to three -point calibration by using two to three buffer solutions that bracket the expected pH range of the ground water, including acidic, neutral and basic solutions. If field measurements fall outside the calibrated range, then the meter may need to be recalibrated with appropriate solutions. Calibration of DO meters should be performed at least once daily and at a greater frequency if significant changes in elevation or atmospheric pressure occur. All calibration and recalibration checks should be recorded in a field notebook or on field forms (Wilson, 1995).

4.5.2 Volumetric Purging & Sampling

Removal of sitting well water via purging is an important step in well sampling. Water remaining in the pump and tubes are likely stagnant prior to purging, and need to be replaced with fresh, representative ground water. Additionally, drawing water through the purging equipment prior to taking stabilization measurements tends to condition the equipment (i.e., remove entrained gas bubbles).

Traditionally, a sample has been collected after purging of a specified volume of water. The various types of sampling and purging equipment, pros and cons, and recommended uses are described in detail in the section on types of equipment (Section 3.2, Types of Equipment). It is recommended that sampling equipment be dedicated to specific wells to eliminate the need for decontamination. This is most important when pumps are used because their intricate design can often make adequate cleaning difficult.

The amount of water purged is usually three to five well volumes. One well volume can be calculated as follows:

V = H x F where:

    V = one well volume.
    H = difference between depth of well and depth to water (ft).
    F = factor for volume of 1-foot section of casing (gallons).

Table 10.6 provides F for various casing diameters. Multiplying the computed volume (V) times the number of desired volumes to be purged will give the volume of water in gallons to be evacuated.

Field stabilization parameters, as discussed above, should be monitored for stability to determine if additional purging is necessary.

At a minimum, at least 3 well volumes should be removed, and the indicator parameters have stabilized. For volumetric purging, it is suggested that stabilization parameters be collected every ½ well volume after an initial 1 to 1½ well volumes are purged (Yeskis and Zavala, 2002). The volume removed between readings can be adjusted as well-specific information demonstrates that the well stabilizes with less volume removed. However, the volume removed must be more than the volume of the pump and sampling tube capacity or the flow through cell (if used), whichever is greater.

Purging should be at or below rates used for development and those observed for well recovery. Excessive rates may result in the introduction of ground water from zones above or below the well screen, which could dilute or increase contaminant concentration in samples. Overpurging also may cause formation water to cascade down the screen, enhance the loss of VOCs, and introduce oxygen into the subsurface, which may alter water geochemistry and affect chemical analysis. As indicated by Puls and Powell (1992), excessive rates may also lead to increased sample turbidity and the exposure of fresh surfaces capable of adsorbing dissolved metals. If bailers are used for purging, entry and withdrawal to and from the water column should be as slow as possible. Water entrance velocities into bailers may have the same undesirable effects as unacceptably high purging rates (Puls and Powell, 1992).

Monitoring wells should be sampled immediately after purging, unless site-specific conditions preclude it (e.g., if some wells are too low yielding). This minimizes the time for physical and chemical alteration of water in the well casing. Where immediate sampling is precluded, sample collection should begin no later than 24 hours after purging.

Table 10.6 Volume of water in one-foot section of well casing.

Radius (½ Diameter)		F[1]
inches	feet*	gallons
0.75	0.625	0.09
1	0.083	0.16
1.5	0.125	0.37
2	0.1.67	0.65
3	0.25	1.47

[1] F is the volume (in gallons) in a 1-foot section of the well and is computed using:

$$F = 3.14r^2 \times 7.48 \, gal/ft^3 \qquad \text{Equation 1}$$

* Where: r = the inside radius (½ inside diameter) of the well casing (ft).

4.5.3 Low-Flow Purging & Sampling

Low-flow purging, also referred to as low-stress purging, low-impact purging, minimal drawdown purging, or Micropurging®, is a method of well purging that does not require large volumes of water to be withdrawn. The term low-flow refers to the fact that water enters the pump intake with a low velocity. The objective is to minimize drawdown of the water column in the well, avoid disturbance of the stagnant water above the well screen, and draw fresh water through the screen. Purging rates are dependent on stabilizing drawdown. Usually, this will be a rate less than 500 ml/min and may be as low as 100 ml/min; however, the rate should not be based on an arbitrary value

The method is based on the principle that water within the screened zone passes through continuously and does not mix with water above the screen. After drawdown has stabilized and indicator parameters have stabilized, water in the screen can be considered representative of water in the formation. Given this, purging of multiple well volumes is not necessary (Kearl et al., 1994; Powell and Puls, 1992; Nielsen and Nielsen, 2002, ASTM Method D6771). A packer assembly may be necessary in fractured bedrock.

Low-flow sampling offers several advantages. It lessens the volume of water to be purged and disposed, reduces aeration or degassing, maintains the integrity of the filter pack, and minimizes disturbance within the well water column and surrounding materials, thus reducing turbidity. Accordingly, filtering of samples

may be avoided, and low-flow sampling may allow for the quantification of the total mobile dissolved phase and the contaminants adsorbed to mobile particles. Disadvantages include higher initial setup costs, need for greater setup time in the field, and increased training needs. In addition, this procedure cannot be sued to sample wells with LNAPL or DNAPL.

When performing low-flow purging and sampling, it is recommended that the pump be set in the center of the well screen interval to help prevent disturbance of any sediments at the bottom of the well. If known, the pump can be placed adjacent to the areas with the highest hydraulic conductivity or highest level of contaminants. However, empirical studies and modeling simulations show that the entire well screen contributes to the sample. Flow into the screen is controlled by the geology near the well, regardless of pump position; higher permeability zones contribute more water (Varljen et al., 2006).

The use of dedicated pumps is preferred to minimize disturbance of the water column. If a portable pump is used, the placement of the pump can increase turbidity and displace water into the formation. Therefore, the pump must be placed far enough ahead of the time of sampling so that the effect of the pump installation has completely dissipated. The time between pump placement and sampling may vary from site to site and may be in excess of 48 hours (Kearl, et al., 1992; Puls and Barcelona, 1996; Nielsen and Nielsen, 2002). Use a submersible pump with an adjustable rate, such as a low-flow centrifugal or bladder pump. The pump should be capable of reducing the flow to 100 ml/min. If using a bladder pump, follow the manufacturer's recommendations for adjusting the emptying/filling cycle to minimize the potential for turbid flow. During subsequent sampling events, try to duplicate as closely as possible the intake depth and the stabilized extraction rate from the previous events.

Because the objective during low-flow purging and sampling is to minimize drawdown, it is important to measure the water level in the well before pumping. To begin purging, the pump should be started at the lowest speed setting and then the speed can be slowly increased until water begins discharging. Check the water level and slowly adjust the pump speed until there is little or no drawdown or drawdown has stabilized. Water level should be monitored and documented frequently during purging; every three to five minutes is recommended. In certain hydrogeologic settings, a well may initially drawdown several feet before it stabilizes. Drawdown can be considered to be stable when it is at or near the asymptotic part of the drawdown curve. Evidence that drawdown is at or near the asymptotic part of the drawdown curve would include less than a total of 0.3 feet difference between at minimum of three consecutive drawdown measurements (Puls and Barcelona, 1996 Reinhart and Smaldone, 2017). The water level should not be allowed to fall to the pump intake level. If the static water level is above the well screen, drawing the water below the top of the well screen should be avoided or minimized as much as possible. To minimize disturbance, pumping rate adjustments are best made within the first fifteen minutes of purging.

A representative sample can be collected when both drawdown and water quality indicators have stabilized. Depending on the well yield, the time to reach stabilization may vary from 15 minutes to over an hour. At least three to five minutes should be allowed to elapse between drawdown and indicator parameter measurements. Otherwise, the ground water sample may be collected before adequate stabilization has occurred. In addition, at least one sampling equipment volume (pump, discharge line and flow-through cell) should be purged before sampling.

Stabilization parameters (see Table 10.5) should be monitored frequently. It is recommended that specific conductance plus two additional parameters be selected. The measurements should be with a hand-held meter or a flow-through-cell and be at least three minutes apart. Measurements should be spaced apart so that the volume of water in the flow through cell or pump and sampling tubes, whichever is greater, is completely replaced.

An indicator parameter can be considered stable when at least three consecutive readings have stabilized. When all selected parameters have stabilized the well may be considered purged and sampling may commence. A turbidity level of less than 10 NTUs is desirable. If the recharge rate of the well is less than the lowest achievable pumping rate, and the well is essentially dewatered during purging, a sample should be taken as soon as the water level has recovered sufficiently to collect the sample, even if the parameters have not stabilized.
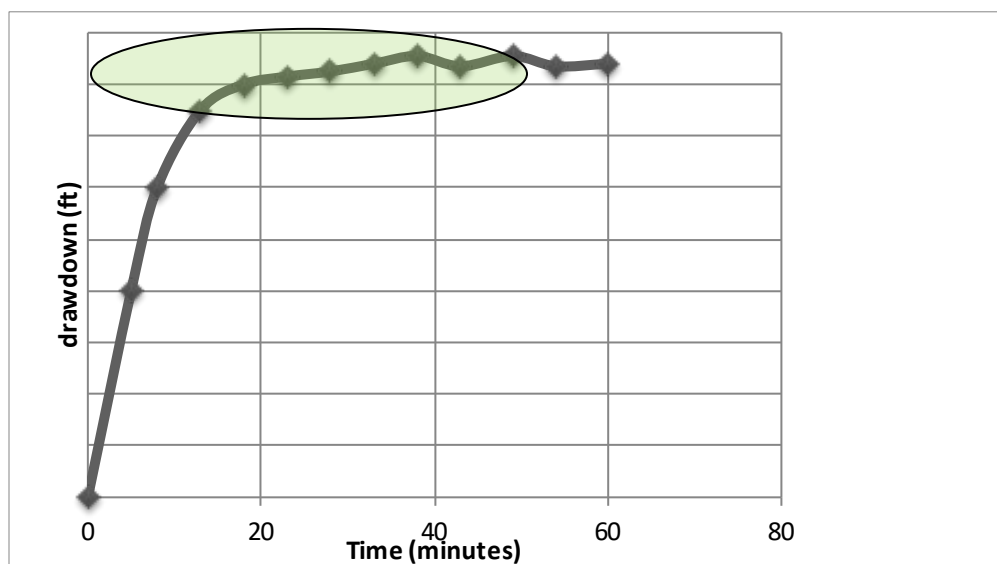
Figure 10.1 Stabilization of drawdown. Shaded area represents area where indicator parameters should be collected to document adequacy of purging.

When conducting low flow sampling at new wells or established wells being sampled for the first time by low flow procedure, it is recommended the purging process be verified by continuing to purge 9 to 15 minutes, then retaking the stabilization parameters. If the parameters remained stable, then the purging procedure can be established for that well based on pump location, rate of purging, and frequency of obtaining the three sets of stabilization parameters. This will help support whether an appropriate amount of water has been purged from the system.

4.5.4 Minimum/No Purge Sampling [10]

Minimum/no purge sampling may be necessary when wells tend to go dry when using other purging and sampling techniques. It involves either pumping just the equipment volume (i.e., pump and discharge tube) prior to obtaining a sample or deployment of a passive sampling device. Minimum/no purge sampling should only be conducted when volumetric or low-flow sampling is not feasible (e.g., well yields less than 100 ml/min). The methodology assumes that the ambient ground water flow through the well screen approximates the ground water chemical conditions within the saturated zone. For low-yielding wells where the flow-through assumption appears to be true, passive sampling techniques are generally considered less disruptive than sampling by purging.

4.5.4.1 Minimum/No Purge by Pumping

This method obtains the sample from within the well screen above the pump intake and removes the least possible volume of water prior to sample collection, which is generally limited to the volume of the sampling system, i.e., pump and discharge tubing. A sample is collected immediately after this volume is withdrawn and is presumed to represent formation water. Very low flow rates are used for minimum/no purge sampling, generally 100 mL/min or less. With minimum/no purge sampling, indicator parameters for chemical stabilization are not monitored. However, indicator measurements may still be needed for other purposes (e.g., regulatory requirements, evaluation of general quality of the ground water).

The volume of water available for sampling within the well screen located above the pump intake should be determined before purging and sampling to avoid drawing stagnant water from the overlying water column into the sample pump and compromising the sample. Drawdown should be measured during pumping to ensure that the water above the screened interval is not drawn into the pump. If possible, the amount of drawdown should be no more than the distance from the top of the screen to the position of the pump intake within the screen, minus a 2-foot safety margin (Figure 10.2) (Nielsen and Nielsen, 2002). In

---

[10]Referred to in some literature as passive sampling.

situations where the 2-foot safety margin above the pump intake cannot be obtained, the volume of water in the pump and discharge tubing at the time the overlying casing water reaches the pump may provide a safety margin.

If available water is insufficient to meet the sample volume requirements, it may be necessary to discontinue the sampling once allowable drawdown is reached. Sample collection should proceed when the well has recharged sufficiently to meet the remaining sampling requirements.

Figure 10.2.   Recommended maximum drawdown for minimum/no purge sampling and purging with a pump.
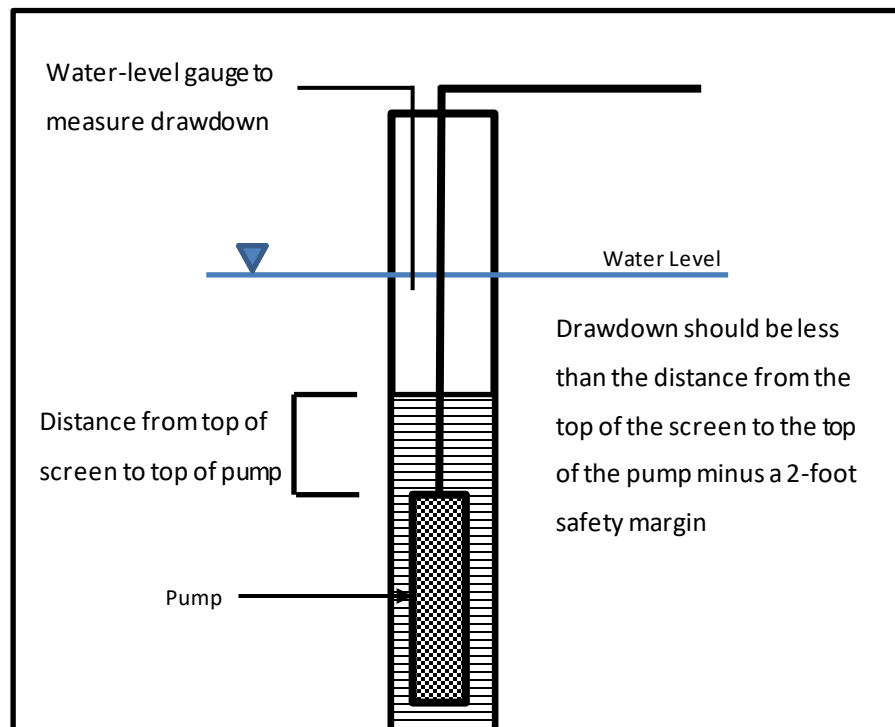
Bladder and low-flow submersible pumps are recommended for minimum/no purge sampling. Pumps should be placed within the well screen, but not too close to the bottom to avoid drawing in any sediments that may have settled, or too close to the top to avoid incorporating stagnant water that is above the well screen. One to two feet above the bottom is generally sufficient. As with low-flow purging and sampling, lowering a pump into the well can increase turbidity and displace water into the formation. Therefore, the pump must be placed far enough ahead of the time of sampling so that the effect of the pump installation has completely dissipated. Though the time between placement and sampling can vary from well to well, it may be in excess of 48 hours (Kearl, et al., 1992; Puls and Barcelona, 1996; Nielsen and Nielsen, 2002).

4.5.4.2 Minimum/No Purge with Passive Sampling Device

Several passive no purge samplers exist. These samplers obtain discrete samples within the well screen where the samplers are placed or activated, rather than a flow-weighted average of the screen zone (unless ambient mixing occurs). Therefore, several samplers may need to be deployed at various depths in the well intake to identify the water quality and any contaminant stratification within the water column. Types of minimum/no purge samplers include the following: diffusion samplers, equilibrated grab samplers and accumulation samplers.

*A **Passive Diffusion Sampler*** can be used to sample for a variety of VOCs. As described in Section 3.2-Types of Equipment, the passive diffusion bag is suspended in the well at the target horizon by a weighted line and allowed to equilibrate with the surrounding water (typically 2 weeks). The sampler bags are retrieved from the well and the enclosed water is immediately transferred to the sample container. Passive diffusion sampling is recommended only for long term ground water monitoring of VOCs at well-characterized sites (ITRC, 2004). PDS is not applicable for inorganics, where there is vertical flow. See Section 3.2.6 – Passive Diffusion Samplers for more description of the applicability of PDS.

Equilibrated Grab Samplers are deployed into the well in a closed position. The sampler allows the well to return to equilibrium, then the sampler is opened to collect the water. Samples are either transferred to containers at the well head or the sampler is shipped to the laboratory for analysis. Examples of equilibrated grab samples include Hydrasleeve™®, Snap Sampler™, and Kemmerer Sampler (U.S.G.S., 2014).

Accumulation Samplers are deployed over a long period of time and use a medium to capture specific COCs. These types of samplers are generally cost prohibitive and specialty or supplier (few) labs are the only ones that can run the samples for analyses. Some of these samplers only gives mass flux results. Examples of accumulation samplers include AGI Sample Module®, Semi-Permeable Membrane Device, diffusion gradient in thin-film, and ceramic dosimeter (U.S.G.S., 2000).

4.5.5 Purge to Dryness & Sampling
Traditionally, low-yielding wells have been sampled by purging a well dry and obtaining a sample upon sufficient recovery of the well. However, there are concerns when a well is purged dry (Nielsen and Nielsen, 2002; U.S. EPA 2001a), including:

- Cascading water as the well recovers may result in a change of dissolved gases and redox state, thus affecting the concentration of the analytes of interest through oxidation of dissolved metals. In addition, the cascading water can strip VOCs that may be present;

- Stressing the formation may increase sample turbidity by inducing soil fines into the well or stirring up any sediments that may have accumulated at the bottom of the well;

- Draining the water from the filter pack may result in air being trapped in the pore spaces, with lingering effects on dissolved gas levels and redox states; and

- The time required for sufficient recovery of the well may be excessive, affecting sample chemistry through prolonged exposure to atmospheric conditions.

Attempts should be made to avoid purging to dryness; however, in some situations it may be the only feasible method (e.g., low yielding wells, insufficient water column to use minimum/no purge). If an operating facility monitoring program has been historically established on purging to dryness, then for consistency, it may be necessary to continue this practice.

If purging to dryness is unavoidable or inadvertent, then samples should be taken when there is a sufficient amount of water to collect a sample that best represents the ground water quality. Extended recovery times after purging (hours) allow the ground water to equilibrate to atmospheric conditions. In the case of a well with very slow recharge, sample collection may continue for several days. However, sample collection should be attempted at least every 24 hours. Herzog et al. (1988) concluded that the common practice of next day sampling for low yield, slow recovery wells is adequate. The intervening time should be consistent from event to event. In addition, it is important to evaluate all data from slowly recovering wells based on the possibility that it may be unrepresentative of actual conditions.

4.6 Filtration[11]
Ground water samples collected from monitoring wells may contain noticeable amounts of sediment. This sample "turbidity" is an important field concern for samples to be analyzed for metals (e.g., cadmium, nickel, zinc) or metalloids (e.g., arsenic, selenium). If large, immobile particles to which metals are bound

---

11 Filtration is banned at RCRA subtitle D facilities (Municipal Solid Waste Facilities) CFR 40 258.53(b).

are allowed to remain in field-acidified samples, laboratory "total" analyses will overestimate the true concentration of mobile species because acidification dissolves precipitates or causes adsorbed metals to desorb. Additionally, changes in the relative degree of sedimentation over time (due to changes in well performance, sampling device, or sampling personnel) and space (due to natural hydrogeologic variations) can result in data interpretation difficulties.

Removal of sediment by filtration prior to containerization and acidification also presents problems. The potential for filter clogging, variable particle size retention, filter media leaching, and aeration is well documented (Puls and Powell, 1992; Striggow et al., 2017). Filtration of water samples to be analyzed for VOCs should be avoided as the filtration process may lead to VOC loss (Connecticut DEEP, 2012; Striggow et al., 2017). Also, filtration has the potential to remove particles that may be mobile in certain hydrogeologic environments. As described by McCarthy and Zachara (1989) and Puls et al. (1990), colloidal material (particles less than 10 micron) may be transported large distances. Because of these difficulties, some investigators (Puls and Barcelona, 1989a & b; Kearl et al., 1992; Puls and Powell, 1992; Striggow et al., 2017) have recommended against field-filtering. Further, federal regulations [40 CFR 258.53(b)] for ground water monitoring at municipal solid waste landfills specify that analyses for metals be performed on unfiltered samples.

For sampling at sites that are not municipal solid waste landfills, filtration may be appropriate in some instances, provided it is done properly (Nielson and Nielson; 2006, Striggow et al., 2017). Significant turbidity is sometimes unavoidable, and filtration may be necessary to remove immobile particles. For example, reducing turbidity may be difficult when a clay-rich glacial deposit is monitored. Clay and natural organic matter can attract contaminants and physically retard particle movement. Therefore, particles in ground water may be presumed to be immobile in formations primarily containing natural organic material and clays. Additionally, while unfiltered data generally would be preferred for a risk assessment of the drinking water pathway, filtered data may be used if there is an obvious discrepancy between filtered and unfiltered data or if secondary maximum contaminant levels MCLs are exceeded (U.S. EPA, 1991). In this case, unfiltered samples might be too turbid to represent drinking water. It is recommended that entities work closely with the Agency to define project requirements. The following sections provide Ohio EPA's general recommendations on whether and how to filter.

4.6.1 Deciding When to Filter

Ohio EPA recommends a general framework (Figure 10.3) for making decisions as to whether filtering is appropriate. Adequate monitoring wells and sampling techniques that minimize disturbance should be confirmed before any decision is made. Filtration generally should occur only when all of the following conditions are present:

- ***The samples have been collected from monitoring wells that are properly designed, installed, and developed.*** Adequate wells are essential to minimizing turbidity and obtaining representative samples. When turbidity is an issue at an existing well, the well should be redeveloped using appropriate well development techniques outlined in Chapter 8 prior to sampling.

- ***The samples have been collected using procedures that minimize disturbance.*** Low-flow purging and sampling procedures are recommended to minimize agitation of the water column and minimize turbidity. Achieve stabilization of indicator parameters prior to sampling to ensure that the sample is representative of natural ground water conditions.

- ***Turbidity has been demonstrated to stabilize above 10 NTU***. (See Section 4.5 – Purging and Sampling Procedure)

- ***Professional judgment indicates that the formation sampled does not exhibit a high degree of particle mobility, and therefore it is reasonable to assume that a portion of the sediment in the samples is likely attributable to immobile particles.*** In general, this judgment can be based on the geology of the ground water zone. For example, clays, because the size of the pores, would prevent particle mobility. Examples of formations that do show significant particle mobility include, but are not limited to, karst; bedrock with open, interconnected fractures, and clean, highly porous gravel-to-boulder sized deposits.

Note that one should exercise professional judgment when applying this approach. Deviations may be necessary if the practices would cause undesirable problems in data interpretation. For example, if a site is underlain by karst bedrock and the historical data for metals has been based on analyses of filtered samples, filtration could be continued to ensure data consistency and comparability. If a single zone is monitored both by wells that are capable of providing samples that meet the turbidity criterion and wells that are not capable of meeting it, it may be prudent to filter all of the samples to ensure spatial consistency and valid statistical comparisons.

Some may wish to collect both filtered and unfiltered samples. The advantage of having both is that a comparison can help determine the form in which a chemical exists (e.g., primarily adsorbed to particulate matter or dissolved) (Puls and Barcelona, 1989b). For example, if the concentration of a chemical is much greater in unfiltered samples compared to filtered samples, it is likely that the majority of the chemical is adsorbed onto particulate matter and not dissolved in the ground water. The comparative data may help justify which data set is more appropriate.
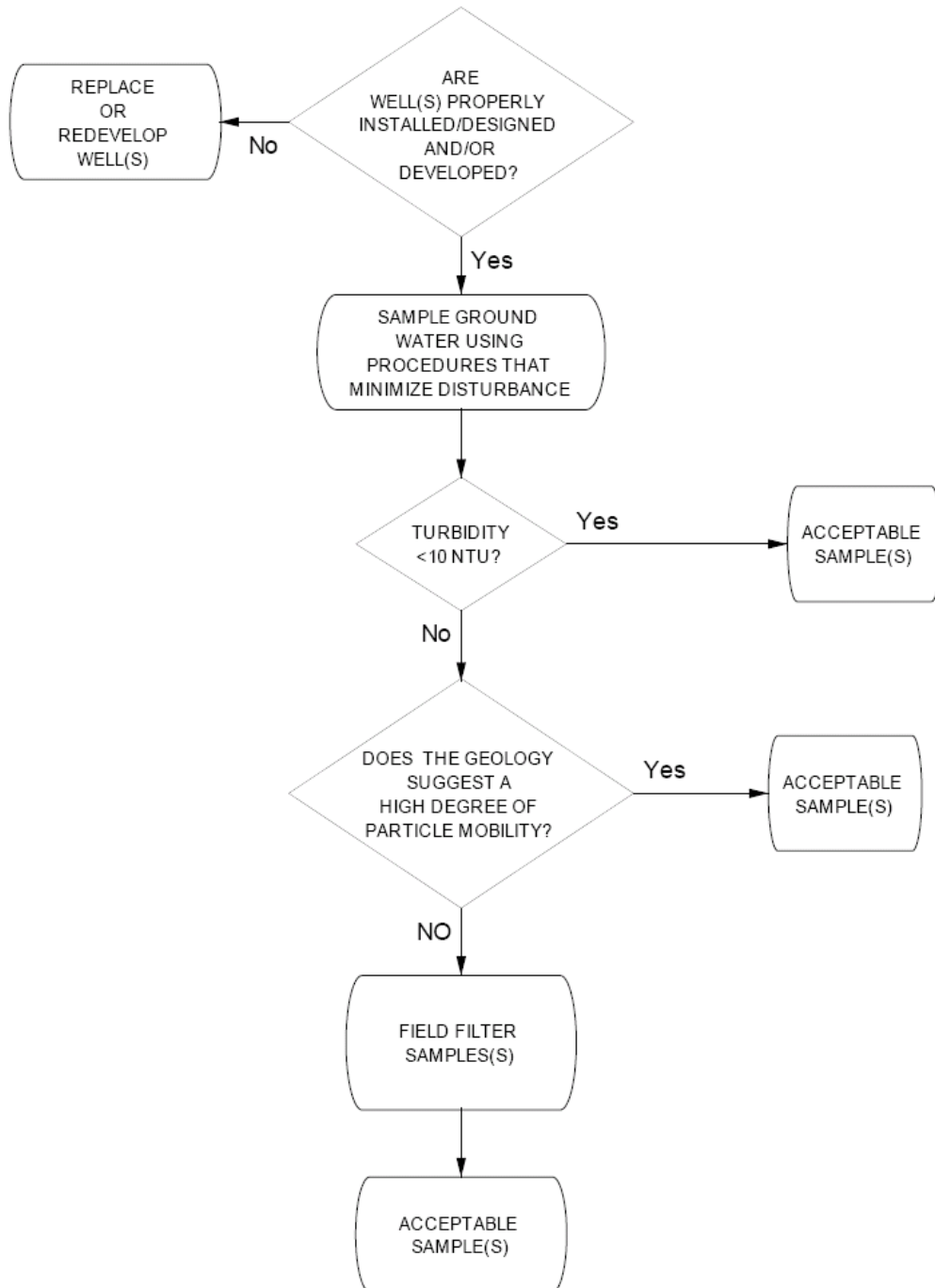
Figure 10.3 Ground water field filtration decision tree

4.6.2. Recommended Procedure/Equipment When Filtering is Necessary

If filtration is necessary, the following are recommended:

- ***Use "in-line" filtering whenever possible.*** In-line methods use positive pressure provided by a sampling pump to force the sample through an attached filter. The advantage is that samples remain isolated prior to atmospheric exposure. Stolzenburg and Nichols (1986) compared different filtering methods and found in-line to provide the best results. If bailers are used for sampling, in-line filters cannot be used unless a pressure or vacuum hand pump (i.e., peristaltic) is utilized to force the

sample through.

- If it is not possible to filter in-line, "open system" techniques may be used. These techniques require a transfer of the sample before filtration, thus allowing for additional exposure and agitation. Open system filtration should be conducted immediately in the field, at the wellhead, and prior to sample acidification and containerization. If filtration does not occur immediately, metals can begin to precipitate and, upon filtration, be removed, causing laboratories to underestimate actual concentrations. Agitation should be kept to a minimum, and the use of "double" filtration is not recommended. "Double" refers to filtering a sample twice using filters with progressively smaller pore sizes. This has been used to speed up filtration; however, it can cause excessive agitation.

- Open system techniques offer varying degrees of portability and ease of decontamination. In addition, changes in pressure and aeration/oxygenation can alter sample representativeness. Open system filtration is primarily driven by either pressure or vacuum mechanisms. For pressure, only pure, inert gas should be used (i.e., nitrogen). If a pump is used, the peristaltic is commonly employed. Whereas pressure "pushes" the sample using compressed gas or a pump, vacuum "pulls" the sample through the filter. Vacuum can cause extensive degassing, which can seriously alter metals concentrations (U.S. EPA, 1986a; EPRI, 1987; and Barcelona et al., 1985); therefore, vacuum is not recommended. The extensive alteration is due to an exacerbation of the pressure decrease inherent with bringing a sample to the surface.

- *Filter samples using a polycarbonate or cellulose acetate filter.* Filtration media should be inert and selected to minimize bias. Polycarbonate membrane filters are recommended. Puls and Barcelona (1989b) have stated that this material should be used due to its more uniform pore size, ease of cleaning, and minimization of adsorptive losses. The NCASI (1982) found polycarbonate to be most appropriate. Cellulose membranes and glass microfiber filters have been used commonly.

- *Prepare the filter prior to collecting the sample.* Filters must be pre-rinsed following manufacturer's recommendations to remove the residue from the manufacturing, packing, or handling. In-line filters should be flushed with sample water before collection to create a uniform wetting front.

- *Use of a 5-micron filter is recommended to ensure that the mobile fraction of turbidity is sampled.* While a 5-micron size filter is recommended, a filter with a different pore size may be used based upon site conditions. Theoretically, the filter pore size should equal the size of the largest mobile particles in the formation, although differences in particles passing different sizes may be lessened significantly by clogging. Traditionally, 0.45-micron filters have been used; however, different pore sizes can be used in specific instances if justified. Puls and Powell (1992) suggested a coarse filter size such as 5-micron. If estimates of dissolved metal concentrations are desired, use of 0.1-micron filters is recommended (Puls and Powell, 1992). Samples filtered with a medium with a small pore size (e.g., 0.1 micron for dissolved concentrations) may be appropriate for geochemical modeling (Puls and Powell, 1992).

- *The filtration medium should be disposed between wells, including filter and syringe.*

- *If the ground water is highly turbid, periodic filter changes may be necessary (e.g., for the same sample)*

- *The filtration device, tubing, etc. should be appropriately decontaminated as sample-contacting equipment (see Section 4.10 - Decontamination)*

4.7 Sample Handling, Preservation, Containers, and Shipping

Once a sample has been removed from a well, appropriate procedures should be utilized to containerize, preserve, and transport it to the laboratory. This ensures that an in-situ state is maintained as much as possible prior to analysis. Issues that should be considered include preservation, containers and labels, holding times, and shipping. Examples of these for some chemicals are listed in Table 10.7. Deviating from Table 10.7 does not necessarily mean that a sample is invalid. Deviations should be recorded on the data reports and should be evaluated on a case-by-case basis, with appropriate reasoning. Appropriate preservation and handling should be coordinated with the laboratory prior to each particular sampling

event.

4.7.1 Sample Acquisition and Transfer

Transfer to a container or filtration device should be conducted in a way that minimizes agitation and aeration. Samples should be transferred directly to the final container for laboratory submittal and not collected in a larger container with subsequent transfer to smaller containers. (Exceptions for filtration are allowable.)   Care should be taken to prevent overfilling so that the preservative, if used, is not overly diluted. If no preservative is used, the containers should be rinsed with sample water prior to collecting the sample. After sealing, containers should not be opened in the field for any reason.

Special considerations are needed when sampling for VOCs. Collection, handling, and containerization should not take place near a running motor or any type of exhaust system. Sample vials should be filled and capped in a manner that minimize bubbles and does not agitate or aerate the sample. Samples should be placed in 40 ml glass vials until a meniscus is formed. Flow rate into the vials should be between 100 and 500 ml/min. The vials should be sealed with a fluorocarbon-lined cap.

The presence of air bubbles in a vial may indicate either improper sampling technique or a source of gas evolution with the sample. Check for air by inverting the vial and tapping. If any bubbles are present greater than 0.25 inches (6 mm) in diameter, the sample should be discarded and a new sample taken (U.S. EPA, 1996b; Yeskis and Zavala, 2002). The container should not be opened and "topped-off" to fill the additional head space (U.S. EPA, 1992). If a sample cannot be obtained without air bubbles due to off-gassing, then the presence of air bubbles should be noted on the field log or field data sheet. Also, air bubbles may form during shipment to the laboratory. These bubbles do not necessarily invalidate the sample[12].

Sampling flow rates "less than 500ml/min are appropriate." (Puls and Barcelona, 1996). Use rates at or below the purging flow rate for metals and other inorganic parameters, lower rates (100 ml/min.) for VOCs and filtered samples. The sample order is dependent on the method of sampling. When volumetric purging/sampling, sampling with bailers, purging to dryness, or no purge sampling is conducted, samples should be collected and containerized in the following order of volatilization sensitivity (U.S. EPA, 1986a):

1. Volatile organic compounds (VOCs).
2. Purgeable organic carbon.
3. Purgeable organic halogens.
4. Total organic halogens (TOX).
5. Total organic carbon (TOC).
6. Extractable organics.
7. Total metals.
8. Dissolved metals.
9. Phenols.
10. Cyanide.
11. Sulfate and chloride.
12. Nitrate and ammonia.
13. Radionuclides.

---

[12]Studies conducted by U.S. EPA indicate that "pea-sized" bubbles (1/4 inch or less in diameter) did not adversely affect data. These bubbles were generally encountered in wastewater samples.

When samples are collected by low flow methods, larger bottles can be filled first, then reduce the flow rate for VOCs and any filtered parameters to 100 to 500 ml/min. The bottles should be filled in a manner that minimizes agitation/aeration.

In addition to the sensitivity, the relative importance of each parameter should be evaluated on a site-by-site basis to establish sampling order protocol. Therefore, when a low-yielding well is being sampled, it may be necessary to change the order of sampling to ensure that a representative sample is collected for the most important constituents for a particular site.

4.7.2 Sample Splitting

Samples are often split into two separate portions and submitted to different laboratories to determine the accuracy of lab results. The proper procedure is to fill the two containers alternately until both are filled. However, if samples for VOC analysis are being collected, the first container should be completely filled, followed by filling of the split container.

4.7.3 Sample Preservation

Preservation is an important step that should be conducted to stabilize the collected sample and prevent physical and chemical changes from occurring during transport to the laboratory and storage before analysis. Preservation is intended to retard biological action, prevent hydrolysis of chemical compounds and complexes, and reduce volatility of constituents (U.S. EPA, 1982). Preservation methods generally are limited to pH control, chemical addition, refrigeration, and protection from light. Appropriate techniques (see Table 10.7), should be implemented immediately upon collection (and after filtration) to minimize changes that begin when a sample is exposed to the atmosphere. Any preservation used should be reported to the appropriate regulatory agency when submitting analytical results.

Sample preservation usually involves reducing or increasing the pH by adding an acid or a base. For example, acids are added to samples submitted for dissolved metals analysis because most metals exist in the dissolved state at low pH. If not preserved, most metals will oxidize and precipitate, which prevents representative analysis. If preserved in the field, the chemical preservative should be obtained from the laboratory contracted to analyze the sample and the appropriate aliquot placed in the sample container, preferably before entering the field. Many laboratories will provide sample bottles containing the appropriate amount and type of preservative. Sampling personnel may want to carry limited amounts of some preservatives in the event that additional preservation is needed for a particular sample. However, if previous samples indicate that a sample may be acidic or alkaline, the amount of preservative should be discussed with the laboratory prior to sample collection.

Samples for temperature-sensitive parameters should be preserved immediately after collection by placement into an insulated cooler maintained at a temperature of approximately 4 - 6º C with ice. Ice substitutes should be avoided. Any deviation in temperature should be noted and assessed as to its impact on sample quality. Care should be taken to ensure that the paperwork and samples are not damaged by ice water. The laboratory should record whether or not the cooler contains any amount of visible ice. The presence of ice is sufficient to demonstrate that the samples are adequately preserved. If no ice is present, the laboratory should obtain a measure or estimate of the sample temperature upon receipt of the samples. [13] This can be accomplished by either a temperature blank or measuring the internal temperature of the cooler.

---

[13]Some regulatory programs may require that the temperature of the cooler/sample be recorded regardless of whether there is visible ice.

Table 10.7 Common Examples of Containers, preservation, and holding times.

PARAMETER	CONTAINER	PRESERVATIVE[2]	MAXIMUM HOLDING TIME
INORGANIC TESTS			
Acidity	P,G	Cool, ≤6 °C	14 days
Alkalinity	P,G	Cool, ≤6 °C	14 days
Ammonia	P,G	None	7 days
		Cool, ≤6 °C; H_2SO_4 to pH<2	28 days
Bromide	P,G	None required	28 days
Chloride	PG,	None required	28 days
Chlorine, residual	P,G	None required	Analyze immediately (within 15 minutes)
Cyanide, total	P,G	Cool ≤6 °C; NaOH to pH>12 ascorbic acid if oxidants (e.g., Chlorine is present.)	14 days
Hardness	P,G	HNO3 to pH<2; H_2SO_4 to pH<2	6 months
Kjeldahl and organic nitrogen	P,G	none	7 days
		Cool, ≤6 °C; H_2SO_4 to pH<2	28 days
Nitrate	P,G	Cool, ≤6 °C	48 hours
Nitrate-nitrite	P,G	Cool, ≤6 °C; H_2SO_4 to pH<2	28 days
Sulfate	P,G	Cool, ≤6 °C	28 days
Sulfide	P,G	Cool, ≤6 °C, add zinc acetate plus sodium hydroxide to pH > 9	7 days
Sulfite	P,G	None required	Analyze within 15 minutes
Metals, except Cr(VI) & Hg	P,G	HNO_3 to pH<2 at least 24 hours prior to analysis	6 months
Chromium (Cr) VI	P,G	Cool, ≤6 °C	24 hours
Chromium (Cr) VI	P,G	use sodium hydroxide and ammonium sulfate buffer solution to pH 9.3 to 9.7 to extend holding time to 28 days	28 days
Mercury (Hg)	P,G	HNO_3 to pH<2	28 days
ORGANIC TESTS			
Volatiles	G, Teflon-lined cap	Cool, ≤6 °C; 0.008% $Na_2S_2O_3$[3] ; HCl to pH<2 No head space	14 days
(Acrolein and acrylonitrile	G, Teflon-lined septum	Cool, ≤6 °C; 0.008% $Na_2S_2O_3$[3], adjust pH to 4-5	14 days
Dioxins and Furans	G, Teflon-lined cap	Cool, ≤6 °C	30 days until extraction, 45 days after extraction
Oil and grease	G	Cool, ≤6 °C; H_2SO_4 or HCl to pH<2	28 days
Phenols	G, Teflon-lined cap	$Na_2S_2O_3$	7 days until extraction, 40 days after extraction
PCBs	G, Teflon-lined cap	Cool, ≤6 °C	1 year
Pesticides	G, Teflon-lined cap	Cool, ≤6 °C; pH 5-9	1 year
Per- and Polyfluorinated Alkyl Substances (PFAS)	250 ml polypropylene bottle	Cool, ≤10 °C; 5.0 g/L Trizma to pH = 7.0	14 days until extraction, 28 days after extraction
RADIOLOGICAL Alpha, beta, and radium	P,G	HNO_3 to pH<2	6 months

[1] Polyethylene (P), Glass (G)

[2] For some constituents the free chlorine must be removed by the appropriate addition of $Na_2S_2O_3$.
Note: The preservative and holding times may vary with sampling procedures, method analysis and selected laboratory. The table is partially based on U.S. EPA, Federal Register, 40 CFR Volume-72, No-47, March 12, 2007 and U.S. EPA Region 3, Sample Submission Procedures for the Office of Analytical Services and Quality Assurance, January 2014.

4.7.4 Containers and Sample Labels

Upon collection, samples should be contained properly to maintain integrity. Specifications on container design, including shape, volume, gas tightness, material construction, and use of cap liners, are defined for specific parameters or suites of parameters. For example, various fluorocarbons (i.e., Teflon®), polyethylene plastic, or glass bottles with Teflon-lined lids are recommended for metals analysis. Samples to be analyzed for VOCs should be containerized in 40 ml glass vials. Specifications on containers are documented in parameter-specific analytical methods (e.g., SW-846). Clean containers can usually be obtained from the contracted laboratory. Note that analytical laboratories may not accept samples for analysis if the bottles have not been cleaned by their own laboratory. If cleaning is necessary, decontamination should be performed, and appropriate blanks collected to verify cleanliness.

Samples should be properly identified with labels. The labels should be permanent and remain legible when wet. When sampling for VOCs the pen's ink may cause false positives, so labels should be completed and the ink allowed to dry before being affixed to the bottles (Wilson, 1995). The following information should be included:

- Sample field identification number (e.g., well location).
- Name or initials of collector.
- Date and time of collection.
- Place of collection.
- Parameters and method requested for analysis.
- Chemical preservatives used.

4.7.5 Shipping

When samples are to be shipped to a laboratory, an appropriate container should be used to protect and preserve them. Coolers are typically used for containing samples and providing temperature preservation. "Wet" ice should always be used for temperature preservation unless otherwise instructed by the laboratory. Ice substitutes such as "blue" ice may cause potential cross contamination issues for some constituents and are not as effective as wet ice for temperature control, especially under warm weather conditions. During warm weather, copious amounts of ice are recommended to avoid having a sample cooler arrive at a laboratory outside of the acceptable temperature range.

Forms such as a sampling request sheet and/or chain-of-custody containing pertinent information should be included (See Section 4.10 Decontamination). Plastic shipping tape should be used to seal sample coolers; do not use duct tape, as commercial shippers cannot accept duct-taped containers due to homeland security concerns. Evidence tape also should be placed around the shipping container (and around each container, if desired), to guard against disturbance or tampering. Special federal requirements apply to hazardous or potentially hazardous samples. For example, flammable, explosive or reactive samples (e.g., LNAPL) cannot be shipped via air carrier (these must be shipped via ground). Always check with the shipping company for packaging, labeling and other requirements or important guidelines. In addition, contact Ohio Department of Transportation (ODOT) and the Public Utilities Commission of Ohio (PUCO) may be contacted for information.

Not all samples will maintain complete stability, regardless of the preservation technique. Therefore, a limit on when analysis should take place has been set for most parameters (see Table 10.7). These "holding times" specify the maximum allowable time between sample collection and laboratory analysis. Depending on the specific circumstances, if one is exceeded, the sample may need to be discarded and a new sample obtained. Therefore, it is important that the time of sampling and transportation to the lab be documented to ensure that the limits are met. Be aware of analytical holding times and minimize the time between sampling and delivery to the laboratory.

4.8 Disposal of Purged Water

Though it is not the intent of this document to define/determine Ohio EPA's policy on disposal of purged

water, the following guidance is provided. If routine sampling shows that the ground water is free from contaminants (e.g., constituent concentrations are not above ambient/natural levels), then it may be acceptable to discharge the purged water onto the ground away from the wellhead but within the limits of the site/facility.[14] Otherwise, the purged water should be containerized until the ground water samples are analyzed. If the water has been contaminated by a listed hazardous waste constituent or exhibits a characteristic of hazardous waste as specified in 3745-51 of the Ohio Administrative Code, it will need to be managed as a hazardous waste. However, if the ground water is treated such that it no longer contains hazardous waste, the ground water would no longer be subject to regulation. Information on this subject can be found at *epa.gov/correctiveaction/resource/guidance/remwaste/refrnces/12cntdin.pdf*.

If the ground water is known or suspected to contain VOCs, air-monitoring equipment may be needed to determine potential exposure and the level of protection for the sampling team. This should be addressed in a site-specific Health and Safety Plan.

4.9 Decontamination Procedures

If non-dedicated sampling equipment is used, it should be cleaned between wells to prevent cross-contamination. This includes all non-dedicated equipment that is submerged in a monitoring well or otherwise contacts a ground water sample. The level of decontamination is dependent on the level and type of suspected or known contaminants. A sampling event where high levels of contaminants are known or suspected would require the most intensive decontamination procedure., However, acid and solvent rinses are not necessary and should not be used, as these procedures create added health and safety concerns for the samplers, may damage sampling equipment, generate hazardous waste and significantly increase the time and level of effort for equipment decontamination. If sampling equipment is contaminated so badly that it cannot be decontaminated using the procedures in Table 10.8 (e.g., a PVC bailer contaminated with LNAPL, Ohio EPA recommends that the equipment be discarded or used only for sampling or removing highly contaminated media (e.g., designating the contaminated PVC bailer for only purging LNAPL from monitoring wells). To help avoid significant contamination of non-disposable sampling equipment, Ohio EPA recommends the use of disposable sampling equipment in highly contaminated monitoring wells. The decontamination area should be upwind of activities that may contribute dust or other contaminants to the solutions used. The process should occur on a layer of polyethylene sheeting to prevent surface soils from coming into contact with the equipment. The effects of cross-contamination can also be minimized by sampling the least contaminated wells first and then progressing to the more contaminated wells, in addition to cleaning equipment between wells.

Table 10.8 outlines sequences and procedures that should be used (modified from ASTM D5088 and Yeskis and Zavala, 2002). The procedures are based on equipment contact with collected samples. Sample-contacting equipment includes non-dedicated bailers and pumps (i.e., devices used for purging and sampling), sample containers, tubing, downhole field parameter probes, water level probes, non-dedicated filtration equipment, etc. In most instances, a distilled water rinse should be sufficient for field parameter measurement probes that are not lowered into wells. Many items are inexpensive and disposable (i.e., gloves, rope, tubing). Items dedicated to a well or disposed of after each well (i.e., gloves, cord, plastic sheet, bailer) would not need to be decontaminated. These items should be properly discarded[15] and new materials provided for the next well.

---

[14]Under detection monitoring, it may be possible to discharge the purged water without containerizing if historical ground water records indicate that ground water quality beneath the site is similar to the ambient quality. The Division with authority over the site/facility should be contacted for approval of this disposal method.

[15]As discussed in the applicable sampling and analysis plan or equivalent protocol, e.g., a standard operation procedure.

Table 10.8 Decontamination procedure for ground water sampling equipment*.

- Wash with non-phosphate detergent and potable water. Using a pressure washer sprayer filled with soapy water and bristle brush is recommended.

- Rinse liberally with potable water.

- Rinse liberally with deionized/distilled water. For containers, this includes 3+ rinses.

- Air-dry thoroughly before use.

- Return equipment to its clean container or wrap with inert material if equipment is not to be used promptly.

4.10 Documentation

4.10.1 Field Sampling Logbook

A field logbook or field sampling forms should be completed and maintained for all sampling events. It should document the following for each well sampled[16].

- Identification of well
- Well depth
- Static water level depth.
- Presence of immiscible layers and detection method
- Thickness of immiscible layers, if applicable
- Well yield estimate
- Purging device, purge volume and pumping rate
- Time well purged
- Measured field parameters
- Collection method for immiscible layers (if applicable) and identification numbers
- Sampling device used
- Well sampling sequence

- Sample appearance
- Sample odor if present
- Types of sample containers and sample identification numbers
- Preservative(s) used
- Parameters requested for analysis
- Field analysis data and method(s)
- Sample distribution and transporter
- Field observations on sampling event
- Name of sampler(s)
- Climatic conditions (e.g., air temperature, precipitation, and wind conditions)
- Problems encountered and deviations made from the established sampling protocol

4.10.2 Chain-Of-Custody

A chain-of-custody record (Figure 10.4) should be established to provide the documentation necessary to trace sample possession from time of collection to final laboratory analysis. The record should account for each sample and provide the following information (U.S. EPA, 1992):

- Sample identification number
- Printed name and signature of sampler
- Date and time of sample collection
- Sample media (e.g., ground water)
- Identification of well
- Number and types of containers
- Constituents and parameters to be analyzed
- Analytical methods
- Preservatives used
- Field QA/QC samples

- Commercial carrier or courier used to deliver samples to the laboratory
- Printed name and signature of person(s) involved in the chain of possession[17]
- Dates/times samples were relinquished by sampler and received by the laboratory
- Internal temperature of shipping container upon opening at laboratory, if applicable
- Presence/absence of ice (when samples are received at the laboratory)
- Special handling instructions (if any)
- Name of person receiving the sample

---

[16]Items documented on the chain-of-custody do not need to be repeated in the field log.

[17]Including all persons relinquishing the samples and all persons receiving the samples, but excluding the U.S. Postal Service, courier services, or commercial shipping companies.

Figure 10.4 Example Chain of Custody Form

4.11 Field Quality Assurance/Quality Control (QA/QC)

To assure adequate QA/QC in the field, the sampling plan should be followed consistently. To verify if procedures are contaminating ground water samples, a variety of samples and blanks need to be collected and analyzed. If re-sampling is triggered, best professional judgment should be used to decide what type of QA/QC sampling is needed. The following are typical checks:

- **Field Duplicates** - Field duplicates are samples collected as close to each other in time and space as practical at a specific location. Ultimately, upon analysis, both should yield the same results within an acceptable range. Excessive variation could indicate problems with the sampling procedures or analysis. At a minimum, duplicates should be collected at a frequency of one per twenty samples (Yeskis and Zavala, 2002), one per week, and one per sampling event.

- **Trip Blanks** - Trip blanks are generally prepared by the laboratory before entering the field. Containers are filled with analyte-free, distilled, deionized water and sealed. These blanks are taken to the field and handled along with the collected samples, thereby acting as a control sample to determine potential VOC contamination from the containers themselves. Trip blanks should be included in each cooler containing VOC samples. At a minimum, at least one trip blank should accompany each sampling event. Trip blanks are never opened in the field.

- **Equipment Rinsate Blanks** - Whenever non-dedicated sampling equipment is used, equipment rinsate blanks should be collected. An equipment blank is obtained by passing analyte-free, distilled, deionized water through a cleaned sampling apparatus (pump, bailer, filtration gear, etc.) and collecting it in a clean container. This blank is used to assess the effectiveness of the decontamination procedures implemented between sampling locations. Ideally, equipment blanks should be collected after sampling the well(s) that historically show(s) the highest levels of contamination. They should be collected at a frequency of one blank per 20 samples (Yeskis and Zavala, 2002), one per week, and one per sampling event.

- **Field Blanks -** Field blanks are samples of analyte free water poured into the container in the field, preserved and shipped to the laboratory. They are used to assess whether there is a potential for sample contamination from field conditions. Analysis from field blanks cannot be used to adjust sample results.

- Matrix Spike and Matrix Spike Duplicates -   **Matrix Spikes and Matrix Spike Duplicates** – A sufficient volume of sample (typically two or three times the volume of a normal bottle set, depending on the laboratory) shall be collected at a frequency of one in every twenty wells sampled for the laboratory to conduct matrix spike (MS) and matrix spike duplicate (MSD) analyses. A MS is a sample that is spiked by the laboratory with known concentrations of known compounds and subjected to the entire analytical procedure. It is used to indicate the appropriateness of the method for the matrix by measuring recovery and accuracy. A MSD is a duplicate of the MS sample. The purpose of the MSD, when compared to the MS, is to determine method precision. The results of the MS/MSD analysis are to be reviewed to assess sampling and analysis precision and accuracy.

- **Temperature Blank** - A temperature blank may be used to estimate the sample temperature at the time the sample is received by the laboratory (ASTM, D6517).

All duplicates and blanks should be subjected to the same analysis as the ground water samples. The results are used to determine if proper procedures were followed. Blank contamination can result from improper decontamination of sampling equipment, poor sampling and handling procedures, contaminated rinse water or preservatives, or the interaction between sample and container. The concentration levels of any contaminants found should ***not*** be used to correct the ground water data. Blank contamination should trigger a re-evaluation of procedures to determine the source of the problem. Sample results should be normalized/adjusted per control results as part of lab results data sheets.

5.0 Ground Water Sample Analysis

5.1 Selection of Analytical Method

The selection of the method for ground water analysis is determined by the parameters of interest and the purpose of the investigation. Several methods may exist for the same parameter. The selected analytical method should be capable of accurately measuring the constituent of concern in the sample. Some regulatory programs may mandate that the analytical method be U.S. EPA-approved or may suggest a preferred method. Therefore, it is recommended that one check with the regulatory program Ohio EPA staff prior to specifying an analytical method.

There are different methods that are approved by U.S. EPA. The following web sites may be helpful in choosing an appropriate method:

> *epa.gov/osw/hazard/testmethods/* - This web site provides resources relating to "Test Methods for Evaluating Solid Waste, Physical/Chemical Methods", also known as SW-846. This site contains updates about SW-846 and information about the development and approval process of SW-846 methods.

> *epa.gov/fem/methcollectns.htm* - Test Method Collections. This site is maintained by EPA's *Forum on Environmental Measurements* as a service to the environmental analytical community

Environmental data may be reported with a variety of detection or quantitation limits. Detection and quantitation limits are not the same. The detection limit of an individual analytical procedure is the lowest amount of analyte in a sample which can be detected but not necessarily quantitated as an exact value. The detection limit is based more upon the sensitivity of an analytical instrument and will only rarely account for the full range of matrix effects that are normally encountered with environmental samples. The appropriate detection and/or quantitation limit should be discussed with the regulatory program and generally will need to be below a human health and/or environmental standard. Various detection limits associated with environmental samples are discussed in the following Ohio EPA documents:

- Tier I Data Validation Manual for the Ohio EPA, Division of Hazardous Waste Management June 13, 2011

- Practical Quantitation Limits. Division of Solid and Infectious Waste Management. April 24, 2007

- Laboratory and Field Screening Data Review. Division of Emergency and Remedial Response. August 19, 2005

5.2 Laboratory Quality Assurance/Quality Control (QA/QC)

Procedures, methods, and levels of quality control are discussed in various U.S. EPA publications (1979a, 1983, and 1986b). Laboratory QA/QC may include, but may not be limited to, qualifications, performance, matrix effects (e.g., blanks and matrix spikes), documentation, and record reporting. For sites under the CERCLA process, Ohio EPA-DERR (1998) has established set guidelines and specifications for preparing quality assurance project plans. Also, information on QA/QC can be found in DHWM Tier I Data Validation Manual (Ohio EPA, 2011).

To obtain reliable results, appropriate laboratory procedures and methods should be followed. An extensive laboratory QA/QC program ensures the production of scientifically sound, defensible results that can be documented and verified. Whether Ohio EPA review is required depends on the regulatory program involved. For example, submittal of a laboratory QA/QC plan is not required for sites undergoing RCRA closure (Ohio EPA, DERR RCRA Program); however, the owner/operator should demonstrate that the laboratory has a plan that contains the elements listed by U.S. EPA (1986b). A laboratory QA/QC plan should be approved for sites remediated under the CERCLA process (Ohio EPA, DERR program).

An appropriate level of laboratory QA/QC data should be submitted with sample results to allow verification that the samples were properly handled and analyzed. A particular regulatory program may dictate the amount and type of data. All QA/QC data should be kept and made available upon request.

This data may be valuable for explaining outliers and questionable results. However, the laboratory QC results should not be used to alter the sample analytical data. A report on analytical data is incomplete without some verification of laboratory QA/QC.

6.0 References

Aller, L., T.W. Bennett, G. Hackett, R.J. Petty, J.H. Lehr, H. Sedoris, D.M. Nielsen, and J.E. Denne. 1991. Handbook of Suggested Practices for the Design and Installation of Ground Water Monitoring Wells. Environmental Monitoring Systems Laboratory, Office of Research and Development, U.S. Environmental Protection Agency (EPA). Las Vegas, Nevada. EPA/600/4-89/034. (In cooperation with National Water Well Association).

ASTM Method D4448. 2019. Standard Guide for Sampling Groundwater Monitoring Wells. ASTM International, West Conshohocken, Pennsylvania.

ASTM, Method D4750. 2001. Standard Test Method for Determining Subsurface Liquid Levels in a Borehole or Monitoring Well (Observation Well). ASTM International, West Conshohocken, Pennsylvania.

ASTM, Method D5088. 2020. Practice for Decontamination of Field Equipment Used at Waste Sites. ASTM International, West Conshohocken, Pennsylvania.

ASTM, Method D6634. 2014. Standard Guide for the Selection of Purging and Sampling Devices for Ground-Water Monitoring Wells. ASTM International, West Conshohocken, Pennsylvania.

ASTM Method D6517. 2018. Standard Guide for Field Preservation of Ground-Water Samples. ASTM International, West Conshohocken, Pennsylvania.

ASTM, Method D6699. 2016. Standard Practice for Sampling Liquids Using Bailers. ASTM International, West Conshohocken, Pennsylvania

ASTM Method D6771. 2018. Standard Practice for Low-Flow Purging and Sampling for Wells and Devices Used for Ground Water Quality Investigations. ASTM International, West Conshohocken, Pennsylvania

Anastas, M.Y. and H.J. Belknap. 1980. Summary of Available Portable VOC Detection Instruments, PEDCO Environmental. EPA Report No. 340/1-80-010.

Backhus, D.A., J.N. Ryan, D.M. Groher, J.K. MacFarlane, and P.M. Gschwend. 1993. Sampling Colloids and Colloid-Associated Contaminants in Ground Water. Ground Water, Vol. 31, No. 3, pp. 466-479.

Barcelona, M.J., J.P. Gibb, J.A. Helfrich, and E.E. Garske. 1985. Practical Guide for Ground-Water Sampling. Illinois State Water Survey. Champaign, Illinois. Contract Report 374.

Barcelona, M.J., J.A. Helfrich, and E.E. Garske. 1988. Verification of Sampling Methods and Selection of Materials for Ground Water Contamination Studies. In A.G. Collins and A.I. Johnson (editors), Ground-Water Contamination: Field Methods. ASTM STP963 ASTM, Philadelphia, Pennsylvania.

Barcelona, M.J., H.A. Wehrmann, M.R. Schock, M.E. Sievers, and J.R. Karny. 1989. Sampling Frequency for Ground-Water Quality Monitoring. Project Summary. EPA/600/S4-89/032. U.S. EPA Environmental Monitoring Systems Laboratory. Las Vegas Nevada. September 1989. 7 p.

Barcelona, M.J., H.A. Wehrman, and M. D. Varljen, 1994. Reproducible Well-Purging Procedures and VOC Stabilization Criteria for Ground-Water Sampling. Ground Water. Vol. 32, No. 1, pp. 12-22.

Brown, G.E., D.A. DuBose, W.R. Phillips and G.E. Harris. 1980. Response Factors of VOC Analyzers Calibrated with Methane for Selected Organic Chemicals. U.S. EPA, Research Triangle Park, EPA 600/2-81-002.

Brownlow, A.H. 1979. Geochemistry. Prentice-Hall, 498 p.

Byrnes, M.E., M.D. Leydorf, and D.B. Smet. 1994. Field Sampling Methods and Remedial Investigations. Lewis Publishers, Boca Raton, Florida.

Connecticut Department of Energy & Environmental Protection (DEEP). 2012. Use of Filters for Groundwater Sampling. Technical Memorandum and Guidance. Bureau of Water Protection and Land Reuse Remediation Division. May 1, 2012. 24 p.

Dalton, M.G., B.E. Huntsman, and K.Bradbury. 1991. Acquisition and Interpretation of Water-Level Data. In: D. M. Nielsen (editor), Practical Handbook of Ground-Water Monitoring. Lewis Publishers, Inc. Chelsea, Michigan, pp. 367-397.

Deutsch, W.J. 1997. Groundwater Chemistry, Fundamentals and Applications to Contamination. CRC Press, LLC, 221 p.

DuBose, D.A., G.E. Brown, and G.E. Harris. 1981. Response of Portable VOC Analyzers to Chemical Mixtures. U.S. EPA, Research Triangle Park, EPA 600/2-81-110.

Electrical Power Research Institute (EPRI). 1987. Sampling Guidelines for Ground Water Quality. EA-4952, Research Project 2485-1. Land and Water Quality Studies Program, Environmental Division. Palo Alto, California.

EPRI. 1985. Field Measurement Methods for Hydrogeologic Investigations: A Critical Review of Literature. EPRI EA-4301. Environmental, Physical and Chemical Program, Energy Analysis Division. Palo Alto, California.

Fujii, Y., K.H. Harada, and A. Koizumi. 2013. "Occurrence of perfluorinated carboxylic acids (PFCAs) in personal care products and compounding agents." *Chemosphere* 93(3) (2013):538-544. *NCBI.* Web. 19 June 2013. https://www.ncbi.nlm.nih.gov/pubmed/23932147

Gaber, M.S. and B.O. Fisher. 1988. Michigan Water Well Grouting Manual. Ground Water Control Section, Michigan Department of Public Health. Lansing, MI. 83 p.

Garner, S. 1988. Making the most of Field-Measurable Ground Water Quality Parameters. Ground Water Monitoring Review. Vol. 8, No. 3, pp. 60-66.

Gibb, J.A., R.M. Schuller, and R.A. Griffin. 1981. Procedure for the Collection of Representative Water Quality Data from Monitoring Wells. Illinois State Water Survey and Illinois State Geological Survey Cooperative Ground Water Report 7, 61p.

Gibbons, R.D. and M.N. Sara. 1993. Statistical Comparison of Metals Concentrations in Filtered and Unfiltered Ground-Water Samples, in Ground Water Sampling, A Workshop Summary: Dallas, Texas. November 30 - December 2, 1993. U.S. EPA Office of Research and Development, EPA/600/R-94/205, pp. 54-58.

Gillham, R.W., M.J. L. Robin, J.F. Barker and J.A. Cherry. 1983. Ground Water Monitoring and Sample Bias. Department of Earth Science, University of Waterloo, Ontario. API Publication 4367. Washington, D.C.

Heath, R.C. 1987. Basic Ground-Water Hydrogeology. United States Geological Survey Water-Supply Paper 2220, 84p.

Hem, J.H. 1992. Study and Interpretation of the Chemical Characteristics of Natural Water (Third Edition). United States Geological Survey Water-Supply Paper 2254, 264 p.

Herzog, B., L.S.F. Chou, J.R. Valkenburg, and R.A. Griffin. 1988. Changes in Volatile Organic Chemical Concentrations After Purging Slowly Recovering Wells. Ground Water Monitoring Review. Vol. 8, No. 4.

Imbrigiotta, T.E., J. Gibs, T.V. Fusillo, G.R. Kish, and J.J. Hochreiter. 1988. Field Evaluation of Seven Sampling Devices for Purgeable Organic Compounds in Ground Water. In: A. G. Collins and A. J. Johnson (editors), Ground Water Contamination: Field Methods, ASTM STP 963. American Society for Testing and Material Standards. Philadelphia, Pennsylvania, pp. 258-273.

Izraeli, R., D. Yeskis, M. Collins, K. Davies, B. Zavala. 1992. Monitoring Well Development Guidelines for Superfund Project Managers. Ground Water Forum. U.S. EPA. Office of Solid Waste and Emergency Response.

Interstate Technology & Regulatory Council (ITRC). 2004. Technical and Regulatory Guidance for Using Polyethylene Diffusion Bag Samplers to Monitor Volatile Organic Compounds in Groundwater. Interstate Technology & Regulatory Council, 39 p.

Karklins, S. 1996. Ground Water Sampling Desk Reference. Wisconsin Department of Natural Resources-Bureau of Drinking Water and Groundwater.

Kasper, R.B. and J. A. Serkowski. 1988. Evaluation of Sampling Equipment for RCRA Monitoring in a Deep Confined Aquifer. Second Outdoor Action Conference, National Water Well Association. Dublin, Ohio. Vol 1, pp. 445-485.

Kearl, P.M., N.E. Korte, and T.A. Cronk. 1992. Suggested Modifications to Ground Water Sampling Procedures Based on Observations from the Colloidal Borescope. Ground Water Monitoring Review. Vol. 12, No. 2, pp. 155-161.

Kearl, P.M., N.E. Korte, M. Stites, and J. Baker. 1994. Field Comparison of Micropurging vs. Traditional Ground Water Sampling. Ground Water Monitoring and Remediation. Vol. 14, No. 4, pp. 183-190.

KEMI. 2015. "Occurrence and use of highly fluorinated substances and alternatives" *Swedish Chemicals Agency.* (2015). *KEMI*. Web. 19 June 2018. https://www.kemi.se/en/global/rapporter/2015/report-7-15-occurrence-and-use-of-highly-fluorinated-substances-and-alternatives.pdf

Kim, K. 2003. Long-Term Disturbance of Ground Water Chemistry Following Well Installation: Ground Water, November-December, 41(6), pp. 780-789.

Lapham, W.W., F.D. Wilde, and M.T. Koterba. 1995. Ground-Water Data Collection Protocols and Procedures for the National Water-Quality Assessment Program: Selection, Installation, and Documentation of Wells, and Collection of Related Data: U.S Geological Survey Open-File Report 95-398, 40 p.

Lee, G.F. and R.A. Jones. 1983. Guidelines for Sampling Ground Water. Journal of the Water Pollution Control Federation. Vol. 55, No. 1, pp. 92-96.

Martin-Hayden, J.M. 2000. Sample Concentration Response to Laminar Wellbore Flow: Implications to Ground Water Data Variability. Ground Water, Volume 38, Number 1, pp 12-19.

McCarthy, J.F. and J.M. Zachara. 1989. Subsurface Transport of Contaminants. Environmental Science and Technology, Vol. 5, No. 23, pp. 496-502.

Muska, C.F., W.P. Colven, V.D. Jones, J.T. Scogin, B.B. Looney, and V. Price, Jr. 1986. Field Evaluation of Ground Water Sampling Devices for Volatile Organic Compounds. In: Proceedings of the Sixth National Symposium and Exposition of Aquifer Restoration and Ground Water Monitoring. National Water Well Association. Dublin, Ohio, pp. 235-246.

National Council of the Paper Industry for Air and Stream Improvement (NCASI). 1984. A Laboratory Study of Four Methods of Sampling Ground Water for Volatile Organic Compounds. Technical Bulletin No. 441.

NCASI. 1982. A Guide to Ground Water Sampling. Technical Bulletin No. 362. New York, New York.

National Ground Water Association (NGWA). 2018. Groundwater and PFAS: State of Knowledge and Practice. NGWA *Press*.

NCASI. 1984. A Laboratory Study of Four Methods of Sampling Ground Water for Volatile Organic Compounds. Technical Bulletin No. 441.

Nielsen, D.M. and G.L. Nielsen. 2006. Ground-Water Sampling In: D. M. Nielsen (editor), Practical Handbook of Environmental Site Characterization and Ground-Water Monitoring. Taylor & Francis Group, CRC Press. Boca Raton, Florida, pp. 959-1112.

Nielsen, D.M. and G.L. Yeates. 2002. Technical Guidance on Low-Flow Purging & Sampling and Minimum-Purge Sampling. Second Edition. The Nielsen Environmental Field School, Inc. Galena, Ohio.

Nielsen, D.M. and G.L. Yeates. 1985. A Comparison of Sampling Mechanisms Available for Small-Diameter Ground Water Monitoring Wells. Ground Water Monitoring Review. Vol. 5, No. 2, pp. 83-89.

Nightingale, H.I. and W.C. Bianchi. 1977. Ground-Water Turbidity Resulting from Artificial Recharge: Ground Water, Vol. 15, No. 2, pp. 146-152.

Ohio EPA. 2011. Tier 1 Data Validation Manual. Division of Hazardous Waste Management. Columbus, Ohio. June 13, 2011. 138 p.

Ohio EPA. 2007. Practical Quantitation Limits. Guidance Document #406. Division of Solid and Infectious Waste Management. Columbus, Ohio. April 24, 2007. 4 p.

Ohio EPA. 2005. Laboratory and Field Screening Data Review. Division of Emergency and Remedial Response.

Ohio EPA. May 2003. 2002 305(b) Report, Ohio's Ground Water Quality. Division of Drinking and Ground Waters. Columbus, Ohio, 78 p.

Ohio EPA. 1990. Guidelines and Specifications for Preparing Quality Assurance Project Plans. Division of Emergency and Remedial Response. Columbus, Ohio.

Oliveros, J.P., D.A. Vroblesky, and M.M. Lorah. 1988. Increasing Purging Efficiency Through the Use of Inflatable Packers. Proceedings of the Second Outdoor Action Conference on Aquifer Restoration, Ground Water Monitoring and Geophysics. National Water Well Association. Dublin, Ohio. Vol 1, pp. 457-470

Parker, L.V. 1994. The Effects of Ground Water Sampling Devices on Water Quality: A Literature Review. Ground Water Monitoring and Remediation. Vol. 14, No. 2, pp. 130-141.

Parsons (Parsons Engineering Science, Inc.). 2003. Final Comprehensive Results Report for the Passive Diffusion Bag Sampler Demonstration Prepared for Air Force Center for Environmental Excellence Technology Division.

Paul, D.G., C.D. Palmer and D.S. Cherkauer. 1988. The Effects of Construction, Installation, and Development on the Turbidity of Water in Monitoring Wells in Fine-Grained Glacial Till: Ground Water Monitoring Review. Vol. 8, No. 1, pp. 73-82.

Pennino, J.D. 1988. There's No Such Thing as a Representative Ground Water Sample (Editorial). Ground Water Monitoring Review. Vol. 8, No. 3, pp. 4-9.

Pohlmann, K.F. and J.W. Hess. 1988. Generalized Ground Water Sampling Device Matrix. Ground Water Monitoring Review. Vol. 8, No. 4, pp. 82-85.

Pohlman, K.F., R.P. Blegen, and J.W. Hess. 1991. Field Comparison of Ground-Water Sampling Devices for Hazardous Waste Sites: An Evaluation Using Volatile Organic Compounds. (Environmental Monitoring Systems Laboratory Office of Research and Development. U.S. EPA, Las Vegas, Nevada. EPA/600/4-90/028. May 1991.

Powell, R.M., and R.W. Puls. 1993. Passive Sampling Ground Water Monitoring Wells Without Purging: Multilevel Well Chemistry and Tracer Disappearance. Journal Contaminant Hydrology, Volume 12, pp. 51-77.

Powell, R.M. and R.W. Puls. (no date). Sampling of Ground Water Monitoring Wells, Part 3-Low Flow and Passive Purging and Sampling. *http://www.powellassociates.com*

Puls, R.W., D.A. Clark, B. Bledsoe, R.M. Powell, and C.J. Paul. 1992. Metals in Ground Water: Sampling Artifacts and Reproducibility. Hazardous Wastes and Hazardous Materials. Vol. 9, No. 2, pp. 149-162.

Puls, R.W. and M.J. Barcelona. 1996. Low-Flow (Minimal Drawdown) Ground Water Sampling Procedure. Ground Water Issue Paper EPA/540/S-95/504., U.S. EPA, Office of Research and Development, Office of Solid Waste and Emergency Response. Washington, D.C. April 1996. 12 p.

Puls, R.W. and M.J. Barcelona. 1989a. Filtration of Ground Water Samples for Metals Analysis. Hazardous Waste and Hazardous Materials. Vol. 6, No. 4, January 1989. pp. 385-393.

Puls, R.W. and M.J. Barcelona. 1989b. Ground Water Sampling for Metals Analyses. Superfund Ground Water Issue, EPA/540/4-89/001. U.S. EPA, Office of Research and Development, Office of Solid Waste and Emergency Response. March 1989. 6 p.

Puls, R.W., J.H. Eychaner and R.M. Powell. 1990. Colloidal-Facilitated Transport of Inorganic Contaminants in Ground Water: Part I. Sampling Considerations. U.S. EPA, Environmental Research Brief. EPA/600/M-90/023. Robert S. Kerr Environmental Research Laboratory. Ada, Oklahoma.

Puls, R.W. and R.M. Powell. 1992. Acquisition of Representative Ground Water Quality Samples for Metals. Ground Water Monitoring Review. Vol. 12, No. 3, pp. 167-176.

Reinhart, R. and J. Smaldone. 2017. Low Stress (low flow) Purging and Sampling Procedure for the Collection of Groundwater Samples from Monitoring Wells, EQASOP-GW4. U.S. EPA Region 1, Quality Assurance Unit. North Chelmsford, Massachusetts. 30 p.

Rose, S., and A. Long. 1988. Monitoring Dissolved Oxygen in Ground Water: Some Basic Considerations. Ground Water Monitoring Review. Vol. 8, No. 1, pp. 93-97.

Schrale, G. and J.F. Brandywyk, 1979. An Acoustic Probe for Precise Determination of Deep-Water Levels in Boreholes. Ground Water. Vol. 17, No. 1, pp. 110-111.

State Coordinating Committee on Ground Water (Ohio). 2015. Regulations and Technical Guidance for Sealing Unused Water Wells and Boreholes, 64 p.

Stolzenburg, T.R. and D.G. Nichols. 1986. Effects of Filtration Method and Sampling on Inorganic Chemistry on Sampled Well Water. Proceedings of Sixth National Symposium and Exposition on Aquifer Restoration and Ground Water Monitoring. National Water Well Association. Dublin, Ohio. pp. 216-234.

Striggow, B., J. Deatrick, and H. Johnson. 2017. Groundwater Sampling. Operating Procedure SESDPROC-301-R4. U.S. EPA Region 4, Science and Ecosystems Support Division. Athens, Georgia. April 26, 2017. 34 p.

Striggow, B., A. Quinones, and L. Ackerman. 2008. Design and Installation of Monitoring Wells. Guidance SESDGUID-101-R0. U.S. EPA Region 4, Science and Ecosystems Support Division. Athens, Georgia. February 18, 2008. 43 p.

Tai, D.Y., K.S. Turner and L.A. Garcia. 1991. The Use of a Standpipe to Evaluate Ground Water Samplers. Ground Water Monitoring Review. Vol. 11, No. 1, pp. 125-132.

Testa, S.M., and D.L. Winegardner. 1991. Restoration of Petroleum-Contaminated Aquifers. Lewis Publishers, Inc., 269 p.

U.S. Army Corp of Engineers. 1998. Monitoring Well Design, Installation, and Documentation at Hazardous, Toxic, and Radioactive Waste Sites: Engineering and Design Engineer Manual EM 1110-1-4000, pp. 6-4.

U.S. EPA Region 3. 2014. Sample Submission Procedures for OASQA Laboratory Branch: Office of Analytical Services and Quality Assurance Laboratory Branch, OASQA Sample Submission Procedures Rev. 13. January 2014. 41p.

U.S. EPA. 2013. Guidelines Establishing Test Procedures for the Analysis of Pollutants Under the Clean Water Act; Analysis and Sampling Procedures. Federal Register, 40 CFR Parts 136, 260, 423, 430 and 435. Wednesday, March 6, 2013. Final Rule.

U.S. EPA. 2009. Statistical Analysis of Groundwater Monitoring Data at RCRA Facilities, Unified Guidance. EPA 530/R-09-007. Office of Resource Conservation and Recovery. March 2009. 886 p.

U.S. EPA. 2007. Guidelines Establishing Test Procedures for the Analysis of Pollutants Under the Clean Water Act; National Primary Drinking Water Regulations; and National Secondary Drinking Water Regulations; Analysis and Sampling Procedure; Final Rule. Federal Register, 40 CFR Part 122. Monday, March 12, 2007.

U.S. EPA. 2006. Guidance on Systematic Planning Using the Data Quality Objectives Process. AQ/G-4. EPA/240/B-06/001. Office of Environmental Information. Washington D.C. 121 p.

U.S. EPA. 2001a. Environmental Investigations Standard Operating Procedures and Quality Assurance Manual. Region 4. Athens, Georgia. 414 p.

U.S. EPA. 2001b. Standard Operating Procedures, Monitoring Well Development. SOP 2044. REV 0.1. Environmental Response Team. October 23, 2001. 8 p.

U.S. EPA. 1997. Proceedings of the Symposium on Natural Attenuation of Chlorinated Organics in Ground Water. EPA/540/R-97/504. Office of Research and Development. Washington, D.C.

U.S. EPA. 1996a. How to Effectively Recover Free Product at Leaking Underground Storage Tank Sites. A Guide for State Regulators. EPA 510-R-96-001. Office of Underground Storage Tanks, Office of Solid Waste and Emergency Response, National Risk Management Research Laboratory. Cincinnati, Ohio. September 1996. 162 p.

U.S. EPA. 1996b. Test Methods: SW846, Chapter 4-Organic Analysis. Revision 3 and subsequent revisions.

U.S. EPA. 1992. RCRA Ground-Water Monitoring Draft Technical Guidance. Office of Solid Waste. Washington, D.C. November 1992. 236 p.

U.S. EPA. 1991. Exposure Point Concentrations in Ground Water. EPA/903/8-91/002. U.S. EPA, Region 3. Office of Superfund, Hazardous Waste Management. Philadelphia, Pennsylvania. 5 p.

U.S. EPA. 1989. Risk Assessment Guidance for Superfund. Volume 1. Human Health Evaluation Manual (Part A). EPA/540/1-89/002. Office of Emergency and Remedial Response. Washington, D.C. 291 p.

U.S. EPA. 1986a. RCRA Ground Water Technical Enforcement Guidance Document. EC-G-2002-130. OSWER 9950.1. Office of Waste Program Enforcement, Office of Solid Waste and Emergency Response. Washington, D.C. 329 p.

U.S. EPA. 1986b. Test Methods for Evaluating Solid Waste, SW846. Third Edition. Office of Solid Waste and Emergency Response. Government Printing Office. Washington, D.C.

U.S. EPA. 1983. Methods for Chemical Analysis of Water and Wastes. EPA/600/4-79/020. Office of Research and Development. Washington D.C. March 1983. 491 p.

U.S. EPA. 1982. Handbook for Sampling and Sample Preservation of Water and Wastewater. EPA-600/4-82-029. (Available through National Technical Information Service, PB 83-124503. Springfield, VA).

U.S. EPA. 1979. Handbook for Analytical Quality Control in Water and Wastewater Laboratories. EPA 600/4-79/019. Environmental Monitoring and Support Laboratory, Office of Research and Development. Cincinnati, Ohio. March 1979. 164 p.

United States Geological Survey (USGS). 2014. Use of Passive Diffusion Samplers for Monitoring Volatile Organic Compounds in Ground Water. USGS Fact Sheet 088-00. URL: ***https://pubs.usgs.gov/fs/fs-088-00/***. (In cooperation with New Hampshire Department of Environmental Services).

Vandenberg, P. and M.D. Varljen. 2000. Low-Flow Purging and Sampling Groundwater at a Solid Waste Landfill: Planning and Implementation. Proceedings of the SWANA 15th Annual Pacific Northwest Regional Symposium. Vancover, B.C. April 2000.

Varljen, M.D., M.J. Barcelona, J. Obereiner, and D.B. Kaminski. 2006. Numerical Simulations to Assess the Monitoring Zone Achieved During Low-Flow Purging and Sampling. Ground Water Monitoring and Remediation. Volume 25, No. 1 pp. 44-52.

Vitale, R.J., O. Braids, and R. Schuller. 1991. Ground-Water Sample Analysis. In David M. Nielsen (editor), Practical Handbook for Ground-Water Monitoring. Lewis Publishers, Inc. Chelsea, Michigan. pp. 501-540.

Vroblesky, D. 2001. User's Guide for Polyethylene-Based Diffusion Bag Samplers to Obtain Volatile Organic Compound Concentrations in Wells. Part 1: Deployment, Recovery, Data Interpretation, and Quality Control and Assurance. USGS Water-Resource Investigation Report 01-4060. Part 2: Field Tests, USGS Water-Resource Investigation Report 01-4061.

Vroblesky, D. and T.R. Campbell. 2001. Equilibration Times, Compound Selectivity, and Stability of Diffusion Samplers for Collection of Ground-Water VOC Concentrations. Advances in Environmental Research – ADV ENVIRON RES. 5. 1-12. 10.1016/S1093-0191 (00)00036-8.

Vroblesky, D. and T. Hyde. 1997. Diffusion Samplers as an Inexpensive Approach to Monitoring VOCs in Ground Water. Ground Water Monitoring Review, Vol. 17, No. 3, pp. 177-184.

Wiedemeier, T.H., M.A. Swanson, D.E. Moutoux, J.T. Wilson, D.H. Kampbell, J.E. Hansen, and P. Haas. 1997. "Overview of the Technical Protocol for Natural Attenuation of Chlorinated Aliphatic Hydrocarbons in Ground Water Under Development for the U.S. Air Force Center for Environmental Excellence". Proceedings of the Symposium on Natural Attenuation of Chlorinated Organics in Ground Water. EPA/540/R-97/504. U.S. EPA. Office of Research and Development. Washington, D.C., pp. 37-61.

Wilde, F.D., D.B. Radtke, J. Gibs, and R.T. Iwatsubo, eds. 1998. National Field Manual for the Collection of Water Quality Data. USGS Techniques of Water-Resources Investigation, Book 9, Handbooks for Water Resources Investigation. Variously paginated.

Wilson, N. 1995. Soil and Ground Water Sampling. Lewis Publishers. Boca Raton, Florida. 188 p.

Yeskis, D. and B. Zavala. 2002. Ground-Water Sampling Guidelines for Superfund and RCRA Project Managers. U.S. EPA, Technology Innovative Office, Office of Solid Waste and Emergency Response. Washington D.C. 53 p.

Appendix A
Additional Information for Sampling Water Supply Wells

Water supply wells are often sampled as part of characterizing a potential pollutant source. This information is helpful for characterizing the extent of a plume and to ensure that the public has a safe source of water.

Many of the same techniques and protocol for sampling monitoring wells also apply to collecting a representative sample from a water supply. This includes the following: planning and preparation; sample preservation, sample containers, handling and shipping; and documentation. These are discussed in other sections of this document. However, there are additional conditions/procedures that must be considered when selecting the sampling point and the actual sample. These additional considerations are summarized below.

Selecting the Sampling Point

The following should be considered when choosing the location to collect a water sample:

- Prior to sampling, existing information such as well construction, yield and depth should be obtained. The Ohio Department of Natural Resources, Division of Water keeps records of all well logs. Well log records can be searched online at ***http://water.ohiodnr.gov/***. If a well log record does not exist, then the local health department should be contacted to see if they have any records. Also, if no log exists, the depth of the well should be measured, if possible, and compared to the ODNR Ground Water Resource maps. These maps can be obtained at the above cited web link.

- The intake of the water supply well should be screened/opened to the targeted ground water zone of interest.

- The tap selected for sample collection should be the closest to the water source and prior to any treatment system. Also, if possible, the sampling point should be prior to entering the residence, office, building, or holding tanks etc. It is noted that for some small systems the first tap downstream from the pressure tank and upstream from any water treatment may be the best tap available.

- The sampling tap should be protected from exterior contamination associated with being too close to a sink bottom or to the ground. Contaminated water or soil from the faucet exterior may enter the bottle during the collection procedure since it is difficult to place a bottle under a low tap without grazing the neck interior against the outside faucet surface. If the tap is too close to the ground for direct collection into the appropriate container, it is acceptable to use a smaller (clean) container to transfer the sample to a larger container. The smaller container should be made of glass or stainless steel, or of the same composition of the sample bottles. Also, if samples are to be collected for bacteria, then the tap needs to be disinfected prior to sampling. The laboratory should provide you with their tap disinfection procedures.

- Leaking taps that allow water to discharge from around the valve stem handle and down the outside of the faucet, or taps in which water tends to run up on the outside of the lip, should be avoided as sampling locations.

- Disconnect any hoses, filters, or aerators attached to the tap before sampling. These devices can harbor a bacterial population if they are not routinely cleaned or replaced when worn or cracked. If disconnection from an aerator, or treatment system, is required, permission should be obtained from the well owner.

- Taps where the water flow is not constant should be avoided because temporary fluctuation in line pressure may cause clumps of microbial growth that are lodged in a pipe section or faucet connection to break loose. A smooth flowing water stream at moderate pressure without splashing should be used. The sample should be collected without changing the water flow. It may be appropriate to reduce the flow for the volatile organic compounds aliquot to minimize sample agitation.

- When sampling for bacterial content, the sample container should not be rinsed before use due to possible contamination of the sample container or removal of the thiosulphate dechlorinating agent (if used). When filling any sample container, care should be taken that no splashing drops of water from the ground or sink enter either the bottle or cap.

Sampling Technique

The following procedures should be followed when collecting samples from water supplies:

1. Ideally, the sample should be collected from a tap or spigot located at or near the well head or pump house and before the water supply is introduced into any storage tanks or treatment units. If the sample must be collected at a point in the water line beyond a pressurization or holding tank, a sufficient volume of water should be purged to provide a complete exchange of fresh water into the tank and at the location where the sample is collected. If the sample is collected from a tap or spigot located just before a storage tank, spigots located inside the building or structure should be turned on to prevent any backflow from the storage tank to the sample tap or spigot. It is generally advisable to open several taps during the purge to ensure a rapid and complete exchange of water in the tanks.

2. If the water system is not actively running, purge the system for at least 15 minutes. Systems that are actively pumped may require less purging (e.g., 3-5 minutes). After purging for several minutes, measure the stabilization parameters (See Section 4.5.1 – Field Measurements of Ground Water Indicator Parameters). Continue to monitor these parameters until three consistent readings are obtained.

3. After three consistent readings have been obtained, samples may be collected. Samples collected from potable water supplies should not be filtered.

A detailed operation/procedural process for sampling water supplies can be found in the following references:

Wilde, F.D., D.B. Radtke, J. Gibs, and R.T. Iwatsubo. 2006. Collection of Water Samples: U.S. Geological Survey Techniques of Water-Resources Investigations, Book 9, Chap. A4. Version 2.0. 231 p.
http://pubs.water.usgs.gov/twri9A4/

U.S. EPA, 2001. Environmental Investigations Standards Operating Procedures and Quality Assurance Manual. U.S. EPA. Region 4. Athens, Georgia. 414 p.
http://www.epa.gov/region4/sesd/eisopqam/eisopqam.html