United States
Environmental Protection
Agency

GROUND WATER RULE
SOURCE ASSESSMENT GUIDANCE MANUAL

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Purpose

The purpose of this guidance manual is solely to provide technical information for water systems and States to assist them in complying with the Ground Water Rule (GWR). The statutory provisions and EPA regulations described in this document contain legally binding requirements. This guidance is not a substitute for applicable legal requirements, nor is it a regulation itself. Thus, it does not impose legally-binding requirements on any party, including EPA, States, or the regulated community. While EPA has made every effort to ensure the accuracy of the discussion in this guidance, the obligations of the regulated community are determined by statutes, regulations, or other legally binding requirements. In the event of a conflict between the discussion in this document and any statute or regulation, this document would not be controlling.

Interested parties are free to raise questions and objections to the guidance and the appropriateness of using it in a particular situation.

Although this manual describes suggestions for complying with GWR requirements, the guidance presented here may not be appropriate for all situations, and alternative approaches may provide satisfactory performance. The mention of trade names or commercial products does not constitute endorsement or recommendation for use.

Authorship

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Contents

Exhibits

List of Acronyms

ASTM	American Society for Testing and Materials
CANVAS	Composite Analytical-Numerical Model for Viral and Solute Transport Simulation
CWS	Community Water System
CDC	Centers for Disease Control and Prevention (U.S. Department of Health and Human Services)
DEM	Digital Elevation Model
DEQ	Department of Environmental Quality
DLG	Digital Line Graph
DMA	Defense Mapping Agency
EPA	United States Environmental Protection Agency
EROS	Earth Resources Observation Systems
ESIC	Earth Science Information Centers
FECTUZ	Finite Element Contaminant Transport - Unsaturated Zone
GIS	Geographic Information System
GPS	Global Positioning System
GPTRAC	General Particle Tracking
GWR	Ground Water Rule
GWS	Ground Water System
GWTT	Ground Water Travel Time
GWUDI	Ground Water Under the Direct Influence of Surface Water
HAV	Hepatitis A Virus
HSA	Hydrogeologic Sensitivity Assessment
IDEQ	Idaho Department of Environmental Quality
ILEPA	State of Illinois Environmental Protection Agency
KYDEP	Kentucky Department of Environmental Protection
MBAS	Methylene blue active substances
MTBE	Methyl tertiary butyl ether
MUIR	Map Unit Interpretations Record
MWCAP	Multiple Well Capture Zone
NAPP	National Aerial Photography Program
NCWS	Non-Community Water System
NCGMP	National Cooperative Geologic Mapping Program
NHDES	New Hampshire Department of Environmental Services
NRC	National Research Council
NRCS	Natural Resources Conservation Service
NSSC	National Soil Survey Center
PWS	Public Water System
RASA	Regional Aquifer-System Analysis
SAR	Source Assessment Report
SCS	Soil Conservation Service
SDWA	Safe Drinking Water Act
SSURGO	Soil Survey Geographic (SSURGO) Data Base
STATSGO	State Soil Geographic (STATSGO) Data Base
SWAP	Source Water Assessment Program
TCR	Total Coliform Rule

USDA	United States Department of Agriculture
USGS	United States Geological Survey
VIRALT	Virus Analytical Transport
WHAEM	Wellhead Analytical Element Model
WHPA	Wellhead Protection Area
WHPP	Wellhead Protection Program
WIDNR	Wisconsin Department of Natural Resources

1. Introduction

1.1 Background

The 1996 Amendments to the Safe Drinking Water Act (SDWA) required the United States Environmental Protection Agency (EPA) to develop national primary drinking water standards requiring disinfection as a treatment technique for all public water systems, including surface water systems, and, as necessary, ground water systems. EPA promulgated the final Ground Water Rule (GWR) on November 8, 2006 as one step in addressing these requirements. (USEPA, 2006a)

The GWR establishes a risk-targeted approach to identify Ground Water Systems (GWSs) susceptible to fecal contamination and requires corrective action to correct significant deficiencies and source water fecal contamination in public GWSs. A central objective of the GWR is to identify the subset of ground water sources that are at higher risk of fecal contamination among the large number of existing GWSs (approximately 147,000), and then further target those systems that must take corrective action to protect public health. This risk-targeted approach includes the following:

- Periodic sanitary surveys of GWSs requiring the evaluation of eight critical elements and the identification of significant deficiencies;

- Triggered source water monitoring of systems that do not achieve 4-log inactivation or removal of viruses;

- Corrective actions to eliminate significant deficiencies and fecal contamination; and

- Compliance monitoring to ensure that disinfection treatment for drinking water is reliably operated where it is used and achieves a 4-log inactivation or removal of viruses.

The risk-targeted approach also includes assessment source water monitoring and hydrogeologic sensitivity assessments (HSAs) as tools that States may use at their discretion to evaluate ground water sources that may be at risk for fecal contamination. Assessment source water monitoring involves the collection of source water samples at regular intervals from ground water sources or wells located in sensitive aquifers, or from wells that are vulnerable to contamination due to other factors determined by the State, and analysis of those samples for fecal indicators. The GWR specifies *E. coli*, enterococci, or coliphage (male-specific or somatic) as suitable fecal indicator organisms. HSAs, described later in this guidance, are determinations of whether GWSs obtain water from hydrogeologically sensitive settings.

Ground water-supplied public water systems (PWSs) are at greater risk of causing waterborne disease if the source ground water is fecally-contaminated and if the finished water does not receive 4-log treatment of viruses (i.e., a 99.99% reduction). The Sanitary Survey Guidance for Ground Water Systems document (USEPA, 2006b) provides additional information regarding the 4-log inactivation requirements.

Public water supplies may transmit fecal contamination if their sources are subject to one or more of the following risk factors: 1) sensitive aquifers; 2) aquifers in which viruses may travel faster and farther than bacteria (e.g., alluvial or coastal plain sand aquifers; 3) shallow unconfined aquifers; 4) aquifers with thin or absent soil cover; 5) wells previously identified as having been fecally-contaminated; and 6) high population density combined with on-site wastewater treatment systems, particularly those in aquifers with restricted geographic extent, such as barrier island sand aquifers. Other risk factors also may allow or facilitate fecal contamination transmission to PWS wells. Assessment source water monitoring should be conducted at PWS wells if any risk factor is identified.

1.2 Purpose and Scope

The purposes of this guidance document are to describe: 1) scenarios when assessment source water monitoring might be advantageous in protecting public health; 2) sensitive aquifers; 3) data sources and methods suitable for use in a HSA and 4) implementation of assessment source water monitoring.

This document does not rank or prioritize the various risk factors that govern source water fecal contamination likelihood. Each of the risk factors listed above, as well as other factors, may be important nationally or in certain locales. This document emphasizes ways to identify readily available information suitable for office, rather than field, determination of risk at an individual PWS well.

The main objective of this guidance document is to identify the most significant risk factors, useful information sources, and simple analytical approaches that may aid State technical staff when considering the need and locations for assessment source water monitoring. Pathogen survival in the water environment and transport through ground water are discussed with emphasis on localities where there is increased likelihood that infectious pathogens may arrive at a PWS well in sufficient number to cause illness. States may elect to perform these assessments based on data availability and existing knowledge about fecal contamination risk in each State.

This guidance emphasizes desktop analytical approaches and associated data sources because a PWS well's aquifer type can be determined without field investigation in most cases. Furthermore, EPA is encouraging States to build upon their source water assessment efforts (i.e., through their Wellhead Protection Programs (WHPPs) and Source Water Assessment Programs (SWAPs)) if they choose to conduct HSAs, and to coordinate efforts among these programs whenever possible.

This document does not address issues related to well construction or sanitary setback distance encroachment. Proper well construction and encroachments into any sanitary setback exclusion zones for existing wells are addressed in the Sanitary Survey Guidance Manual for Ground Water Systems (USEPA, 2006b). Although proper well construction is an important barrier to contamination for wells situated in all aquifer types, proper well construction does not preclude contamination of water drawn into wells. Conversely, poor well construction may put wells at risk of contamination regardless of the aquifer type in which the well is situated. EPA has included information on appropriate well construction and survey of land use around the well head in the Sanitary Survey Guidance Manual for Ground Water Systems (USEPA 2006b). Although EPA recognizes that land use outside any sanitary setback boundary is often not under the control of the utility, States should consider land use in making decisions about the need for assessment source water monitoring.

This document does not address ground water sources that are under the direct influence of surface water (GWUDI). The GWR is not applicable to GWUDI sources. Sources that have been identified as GWUDI are subject to the requirements for surface water supplies (CFR 40, Part 141, Subparts H, P, T and W). If assessment source water monitoring or other investigations, as part of GWR implementation, identify GWUDI sources that have not been previously identified, then evaluations and regulatory determinations based on surface water supplies should be implemented.

As used in this guidance, "State" refers to the agency of the State or Tribal government that has jurisdiction over public water systems. During any period when a State or Tribal government does not have primacy enforcement responsibility pursuant to section 1413 of the Safe Drinking Water Act, the term "State" means the Regional Administrator, US Environmental Protection Agency.

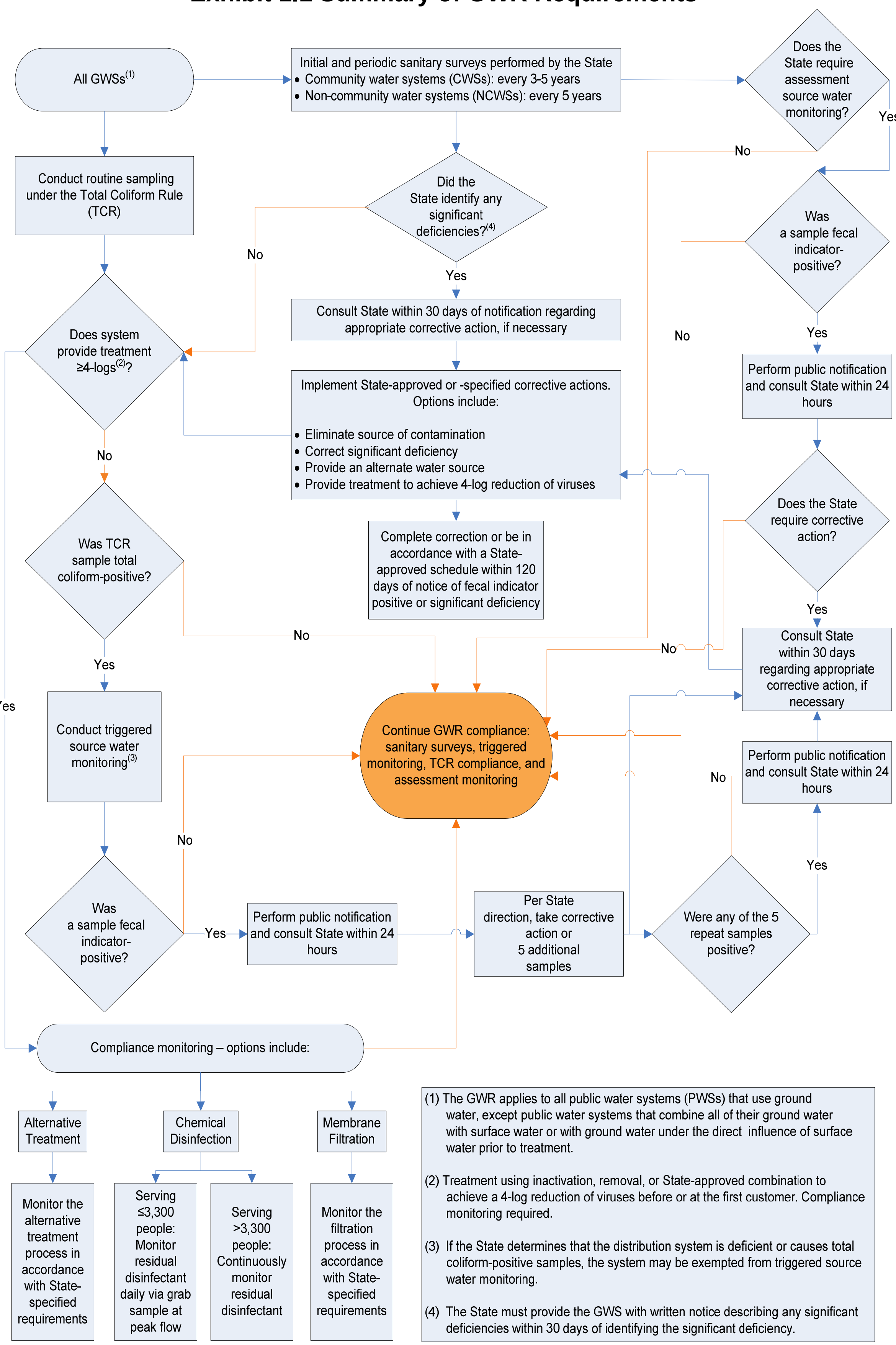
1.3 Ground Water Rule Summary

The GWR applies to all PWSs that use ground water, except PWSs that combine all of their ground water with surface water or with GWUDI prior to treatment. The GWR also applies to consecutive systems receiving finished ground water. Ground water systems (GWSs) must comply with the GWR beginning December 1, 2009. Key components of the GWR are:

1. Sanitary surveys,
2. Triggered source water monitoring,
3. Corrective actions, and
4. Compliance monitoring.

Each of these components is discussed further below and Exhibit 1.1 provides a summary flowchart of the final GWR requirements.

Exhibit 1.1 Summary of GWR Requirements



Sanitary Surveys

The final GWR requires regular (every three years for community water systems (CWSs) and every five years for non-community water systems (NCWSs)) comprehensive sanitary surveys of 8 critical components: (1) source; (2) treatment; (3) distribution system; (4) finished water storage; (5) pumps, pump facilities, and controls; (6) monitoring and reporting, and data verification; (7) system management and operation; and (8) operator compliance with State requirements. If a significant deficiency is identified, corrective action is required or a treatment technique violation is incurred.

Source Water Monitoring

In the final GWR, systems not achieving, or not performing compliance monitoring for, 4-log treatment of viruses (using inactivation, removal, or a State-approved combination of these technologies) must conduct triggered source water monitoring for the presence of at least one of the following fecal indicators: *E. coli*, enterococci, or somatic coliphage. The triggered monitoring requirements apply to systems that are notified that a Total Coliform Rule (TCR) routine sample is total coliform-positive. Within 24 hours of receiving the total coliform-positive notice, GWSs must collect a source water sample and test it for the presence of a fecal indicator.

If the State does not require corrective action (see Corrective Action section below) for an initial fecal indicator-positive source water sample, the system must collect five additional source water samples within 24 hours of being notified of the initial fecal indicator-positive source water sample. The GWR requires systems to take corrective action if any of the five additional source water samples are fecal-indicator positive.

The GWR provides States with the option to require systems to conduct assessment source water monitoring for ground water sources as needed. States may specify the number of samples, the duration of sampling and the analytical method used for analysis. States may use HSAs as a tool to determine if GWSs are obtaining water from hydrogeologically sensitive settings. The GWR requires GWSs to provide the State with any existing information that will enable the State to perform the HSA. States also have the option to require corrective action for any fecal indicator positive sample found during assessment source water monitoring.

Corrective Action

The GWR requires that systems implement corrective action for;

1. Significant deficiencies,

2. Fecal-indicator positive samples, if directed by the State, after the initial fecal indicator-positive in triggered monitoring, or for a fecal indicator-positive found during assessment monitoring, or

3. A fecal indicator-positive sample in any of the five additional source water samples collected after the initial fecal indicator-positive source water sample during triggered monitoring.

The system must implement at least one of the following corrective actions: correct all significant deficiencies; provide an alternate source of water; eliminate the source of contamination; or provide treatment that reliably achieves at least 4-log treatment of viruses. Furthermore, the system is required to notify the public served by the water system of any uncorrected significant deficiencies and/or source water contamination. (The State may also require notification of corrected significant deficiencies.)

Compliance Monitoring

Compliance monitoring requirements are the final defense against microbial contaminants provided by the final GWR. All GWSs that provide 4-log treatment of viruses, either as a corrective action or in lieu of GWR triggered source water monitoring, must conduct compliance monitoring to demonstrate continual treatment effectiveness.

1.4 Public Health Risk Factors

In the GWR preamble, EPA identified several risk factors that may indicate the need for assessment source water monitoring. A PWS well may be at risk if there is an increased likelihood for pathogenic bacteria or viruses to arrive at the well in an infectious state. Because most pathogens are not native to ground water, they are unable to reproduce in the ground water, and their survival is limited. As the subsurface residence time increases, the proportion of infectious pathogens (with that residence time) decreases. Thus, subsurface residence time is often used as a surrogate measure of pathogen risk. An important corollary is that the larger the number of introduced pathogens, the greater the likelihood that some pathogens will remain infectious at any particular residence time. In general, at shallow ground water temperatures, viral pathogens likely remain infectious for a significantly longer time as compared with the bacterial pathogens (Appendix C).

To illustrate this concept, consider a visitor to a highway rest area served by a septic tank. The visitor is recently ill and is shedding pathogens in his stool, which are flushed into a poorly sited and performing septic tank. The risk that others will become ill from drinking water from a nearby PWS well that produces water from a shallow unconfined aquifer and whose water is not receiving 4-log treatment of viruses depends on the time that the pathogens spend in the subsurface. If the average ground water travel time from the aquifer fed by the septic tank is 10 years, then it is likely that all pathogens will be inactivated during their residence in the subsurface, and it is likely that the PWS well is not at risk. If the average ground water travel time is 2 years, then some ground water will take a fast path and arrive in 1 year or less, and other ground water will take a slower path and arrive in 3 years or more. Because pathogens remain infectious in the subsurface for a maximum of about one year (See Appendix C), the health risk depends on the proportion of ground water that arrives most rapidly at the well. That risk increases if the number of pathogens introduced into the groundwater is large rather than small because there is a greater likelihood that the fast arriving ground water will entrain pathogens if there are more pathogens entering the ground water. In any case, the number of infectious pathogens must be sufficiently large (an infectious dose) so as to overcome the innate defense systems in the human host. Viruses have significantly lower infectious doses than do bacteria (McBride et al., 2002).

Exhibit 1.2 Summary of Risk Factors for Targeting Susceptible Systems for Assessment Source Water Monitoring

Risk Factor	Aquifer	Recommended Indicator	Example sources of Information
Sensitive Aquifers	Karst, fractured bedrock, or gravel	*E. coli*, Enterococci, or Coliphage	See section 1.5 and Chapter 2
Aquifers in which viruses may travel faster and further than bacteria	Alluvial or coastal plain sand aquifers	Coliphage	Research literature. See also shallow unconfined aquifers
Shallow unconfined aquifers	Any	Coliphage	Well logs, well construction reports, or well permits
Aquifers with thin or absent soil cover	Any	*E. coli*, Enterococci, or Coliphage	Soil maps. See section 1.5.3
Wells previously identified as having been fecally-contaminated	Any	Based on historical contamination	Sanitary survey records
High population density combined with on-site wastewater treatment system	Barrier island sand aquifers	Coliphage	Sanitary survey records
Other Risk Factors[1]	Any	*E. coli*, Enterococci, Coliphage	Sanitary survey, Source Water Assessment, and field visit records

[1]Including but not limited to: well near a source of fecal contamination; well in a flood zone; improperly constructed well (e.g., improper surface or subsurface seal); well of unknown construction (e.g., no driller's log or other record of construction); other non-microbial indicators of potential for fecal contamination (e.g., Methylene blue active substances (MBAS), high chloride or nitrate levels from baseline or historic trends).

Exhibit 1.3 Outbreak Examples by Risk Scenario

Risk Factor	Example Outbreak
Sensitive aquifers	A norovirus outbreak in 2001 was associated with visitors to a snowmobile lodge in Wyoming (Anderson, et al, 2003). A detailed investigation by Gelting et al (2005) concluded that the bedrock underlying the shallow (coarse-textured) soils at the lodge consisted of fractured granite. When the wastewater reached the fractured granite bedrock, it traveled within the fractures which served as conduits to the PWS wells.
Aquifers in which viruses may travel faster and farther than bacteria	Passengers traveling by bus through Alaska and the Yukon Territory became ill after consuming water at a restaurant in the Yukon Territory (Beller, et al, 1997). Water was supplied to the restaurant by two shallow wells. Wastewater was piped to a septic pit located about 15 m from one well. Well #1 had total and fecal coliform counts of 10-50 per 100 ml and 2-18 per 100 ml. Norovirus was identified in the well water from well #1 and matched to the norovirus recovered from ill individuals. Dye, introduced into the septic pit, arrived at well #1 in about 24 hours.
Shallow unconfined aquifers	A norovirus outbreak is associated with well water at a new resort in Arizona (Lawson, 1991). Although the latest technology was used to design the resort's water and sewage treatment plants, well water was sufficiently contaminated so as to cause illness in about 900 individuals. The waste water passed through about 10 m of sandy alluvium, 70 m of sandstone and contaminated the underlying aquifer. The aquifer is unconfined and is shallow compared with other aquifers in the arid western States.
Aquifers with thin or absent soil cover	At an island in northern Michigan, 39 people became ill from drinking tap water at a resort (Ground Water Education in Michigan, 1992; Chippewa County Health Department, unpublished report, 1992). The septic tank was believed to be the contamination source. Dye introduced into the septic tank was found in the PWS after two days. As described in the outbreak report, much of the island is covered with a thin layer of soil, sometimes insufficient for proper filtration of surface water and sewage effluent.
Wells previously identified as having been fecally-contaminated	An *E. coli* O157:H7 outbreak in Walkerton, Ontario resulted in six deaths and 2300 illnesses (Hrudy and Hrudy, 2004) in 2000. The source water from the wells and from 2 to 6 locations in the distribution system were typically sampled weekly. The available data are summarized in a report by B.M. Ross and Associates Ltd. (2000). In a 1992 report, 3% of 125 samples showed adverse results. In contrast, no adverse results were reported in 1989 and 1991. In 1996, 3% of source water samples and 6% of distribution system samples were adverse. In 1998, 16 adverse samples were identified, and *E. coli* were identified in low numbers (one to four) in treated water and distribution system samples. In 1999, total coliform bacteria were identified in 7 of 151 source water samples and in 3 of 146 distributions system samples. No samples were positive for *E. coli*. In early 2000, prior to the contamination event and outbreak, total coliform bacteria were identified in source water, treated water and the distribution system samples.
High population density combined with on-site wastewater treatment systems	Tourists visiting a resort island in Ohio (and drinking water from PWS wells rather than the centralized water treatment plant) became ill from *Campylobacter*, *Arcobacter*, *Salmonella* and perhaps other pathogens (O'Reilly et al, 2007; Fong et al, in press). The island is served by a centralized water treatment plant but 13 PWS wells and numerous domestic wells were also in use. The island hosts about 25,000 visitors per day during the tourist season. O'Reilly et al (2007) conclude that there was widespread contamination of the aquifer because many wells from various locations on the island showed evidence of contamination.

The following text identifies risk scenarios that may merit additional assessment source water monitoring. These risk scenarios are identified based on the lessons learned from past waterborne disease outbreaks. One common characteristic of the risk scenarios is that all are based on a short ground water residence time (months rather than years) either because the ground water flow is naturally fast or because well pumping combined with short ground water flow paths precludes longer residence times. For example, when fecal indicators are found in PWS well water, that finding is evidence of a short ground water residence time because fecal indicators probably cannot survive in ground water longer than one year outside of a mammalian host (see Appendix C). A short ground water residence time can include both the time to infiltrate from the surface to the aquifer as well as the time exclusively within the aquifer. The purpose of this document is to emphasize issues related to the time within the aquifer as opposed to the infiltration time. Short infiltration times are a feature of GWUDI of surface water wells and are considered in detail in the Surface Water Treatment Rule guidance.

Sensitive aquifers - A sensitive aquifer is herein defined as any karst, fractured bedrock, or gravel aquifer. In these aquifers, ground water flow velocities are typically very high, and flow typically takes the shortest and most direct path. This rapid transport allows fecal contaminants to travel without significant reduction in numbers through inactivation or removal. Furthermore, ground water velocities in these aquifers are even higher in the vicinity of a pumping well than they are under natural flow conditions, increasing the potential for pathogens to migrate to a well.

In addition to karst, fractured bedrock, and gravel aquifers, States may designate additional aquifer settings (e.g., sand and gravel aquifers) as sensitive and require assessment source water monitoring in them if they believe this designation is necessary to protect public health. Sensitive aquifers are discussed in more detail in Chapter 2, including specific methods for identifying sensitive aquifers and case study examples of the process.

As mentioned above, fractured bedrock is defined in this document as a sensitive aquifer. Granite is recognized as an example of fractured bedrock. Wells in granitic and/or similar rocks are able to produce water because there are fractures present and should be considered for assessment source water monitoring to protect public health. This is because unfractured granite is among the most dense and compact rock types and has insufficient connected void space to provide sustained water yield to a well. Thus, to serve as an aquifer, granite must have numerous large, open (unmineralized) and connected fractures which can serve as conduits for contamination. Section 1.5 identifies the hydrogeologic data sources suitable for identifying sensitive aquifers.

Because ground water flow is fast and direct through the relatively large voids within these aquifers, there are few opportunities for entrained fecal indicator microorganisms to interact with the solid aquifer materials. Organism, size, charge or other factors governing transport in ground water are not likely to be significant if there is little interaction with the aquifer solid material. Thus, these organisms will most likely pass through the subsurface at or near the average ground water velocity. All three indicator microorganisms will likely pass through sensitive aquifers at similar rates. Based on source water risk factors, there is no reason to favor one organism over another in selecting a recommended fecal indicator organism.

Aquifers in which viruses may travel faster and farther than bacteria - Aquifers are broadly classified into two categories; porous and non-porous media. A sand aquifer is an example of a porous media aquifer. In a sand or other porous media aquifer, the ground water takes a relatively slow and indirect path around the myriad sand grains, thereby providing ample opportunity for bacteria or viruses, entrained with the water, to come in contact with a sand grain. In a non-porous media aquifer (e.g., fractured bedrock aquifer), the ground water flow can be idealized as fast (laminar) flow between parallel plates. In this flow regime, bacteria and viruses tend to remain within the high velocity zone in the middle and less commonly approach or transition through the boundary layer to attach to the fracture wall.

In this document, non-porous media aquifers and one porous media aquifer, gravel, are defined as sensitive aquifers. Among the remaining porous media aquifers (e.g. sand or sand and gravel aquifers), sand and gravel aquifers (that is, aquifers containing a combination of both sand and gravel) transmit fecal contaminants more efficiently than sand aquifers because average ground water velocity is higher. Assessment source water monitoring in sand and gravel aquifers may be conducted using any of the GWR fecal indicators. Some sand aquifers may also efficiently transmit fecal contamination but, as discussed next, appear to more efficiently transmit viruses as compared with bacteria. Thus, sand aquifers, if targeted for assessment source water monitoring may be monitored using coliphage rather than *E. coli* or enterococci.

Because viruses may travel faster and farther in some aquifers, monitoring in these aquifers using coliphage (virus) may be a better choice than monitoring using bacteria. This issue is also discussed in the GWR Source Water Monitoring Methods Guidance document (USEPA 2006c). Microbial transport in porous media aquifers is an active research area and consensus is difficult. It is generally agreed that microbe size is an important element in determining subsurface transport in porous media (mobility), although many other factors, such as surface charge may also have significant influence. Assuming that size is important, the significant (one-thousand fold) size difference between viruses (measured in nanometers) and bacteria (measured in micrometers) increases the likelihood that an infectious virus rather than an infectious bacterium will reach a PWS well in porous media.

All subsurface particles, including microbes, may be transported by flowing ground water. Particles may permanently (i.e., removal) or temporarily (i.e., retardation) become associated with the solid aquifer materials (both porous and non-porous media). The thousand-fold size difference between viruses and bacteria may be significant in sand aquifers for two reasons: (1) viruses are less likely to be subject to removal or retardation at pore margins by straining, wedging, or micro-straining, and (2) viruses may be more likely to be excluded from the smaller pores where ground water velocities are slower. As a result of this pore-size exclusion (which is due indirectly to size because charge effects predominate for smaller particles), viruses may be favored over bacteria because the viruses remain in faster flowing ground water for longer periods. As a result of straining and pore-size exclusion, sand aquifers may facilitate virus, as compared with bacterial transport.

In other aquifers, such as non-porous media and gravel aquifers where average ground water velocities are exceptionally fast, straining and pore-size exclusion are much less significant and bacteria and viruses are assumed to travel at equal rates. In general, this guidance assumes that straining and pore-size exclusion effects are more significant in sand aquifers where ground water velocity is moderate because mean grain size is moderate (sand aquifers as compared with sand and gravel). As ground water velocities increase because of increasing gravel content or increasing proximity to a pumping well, the differences between virus and bacterial transport efficiency become less important. On the other hand, the finest grained porous media, such as shale and clay beds, are not considered to be important aquifers because water and entrained pathogens are not transmitted efficiently in all directions, so they are not further considered, despite the much greater significance of straining and pore-size exclusion.

Although the presence of viruses in water drawn from a well in porous media implies fecal contamination at or near the surface, some microbial pathogens such as *Legionella pneumophila* (Costa et al., 2005; Riffard et al., 2004), *Helicobacter pylori* (Hegarty et al, 1999; Rolle-Kampcyk et al, 2004), *Naegleria fowlerii* (Blair and Gerba, 2006) and perhaps *Toxoplasma gondii* (Sroka et al, 2006) are not associated with fecal contamination and, instead, may be resident members of aquifer ecosystems. For these microbes, transport from the surface or near surface is not an important risk element because the microbes can colonize the well gravel pack or the aquifer immediately surrounding the gravel pack. In these instances, the bacterial versus viral size difference and associated subsurface mobility become much less important. Municipal waste water that is injected into deep brine formations or aquifer storage and recovery operations are not typically shallow fecal contamination sources and are regulated under the Underground Injection Control provisions of the SDWA. Because they are specifically regulated, they are not further considered here.

In the Alaska outbreak of Exhibit 1.3, norovirus was sufficiently long-lived and mobile in the subsurface to contaminate a well in large numbers sufficient to provide an infectious dose that made visitors ill. In contrast, fecal coliform were probably less mobile and long-lived and arrived at the well only in small numbers. No travelers became ill from bacterial pathogens and no bacterial pathogens were recovered from the well water. The short flow path between the septic pit and the well provided for a relatively short ground water residence time. Wells with fecal contamination sources that are near or at the State-mandated or recommended setback distance should be considered for assessment source water monitoring. Sanitary surveys and source water assessments identify fecal contamination sources present in or near State mandated or recommend setback distances.

There are no simple, desk-top methods to identify PWS wells in which viruses may be more likely than bacteria to arrive at a well. This is an active research area using water and bioparticle tracers and is usually conducted at a field research site or in a laboratory column. However, it is sometimes suggested that PWS wells producing from shallow, unconfined aquifers are examples of PWS wells in which viruses are more likely than bacteria to arrive at a well. This risk factor is discussed next.

Shallow unconfined aquifers - As discussed above, it is commonly assumed that fecal contaminant sources originate at or near the surface. For aquifers that are relatively close to the surface and unprotected by a hydrogeologic barrier such as a confining layer, the transport path is relatively short and unimpeded. Thus, there is a greater likelihood that infectious fecal contamination will reach a PWS well that produces water from a shallow, unconfined aquifer as compared with a deeper or confined aquifer. Shallow aquifers are often, but not always, sand, sand and gravel, or sensitive aquifers. A PWS well producing water from a shallow, unconfined sandstone aquifer has an increased likelihood of infectious fecal contamination as compared with a deep sandstone aquifer. In this instance, it is relatively easy to determine shallow versus deep aquifers because well depth data are always available on the well log, well construction report, or final well permit. If well construction reports are used, it is important to get the well depth from the final report rather than from the well construction permit because many wells are not constructed as designed.

This guidance document does not define which unconfined aquifers are shallow, may be at risk, and therefore are suited for assessment source water monitoring. Aquifer depth is highly variable due to pumping, recharge, geologic structure, and climate, and no single definition applies nationwide. States should consider the range of PWS well depths within their State and geologic provinces and conduct source water monitoring assessment monitoring at those wells producing water from unconfined aquifers that are among the shallowest in the State.

Fecal indicator microorganisms may pass through shallow unconfined aquifers at vastly differing transport rates, depending on the aquifer type. Because groundwater flow and indicator transport is so variable, no fecal indicator is favored and any of the three recommended indicators is appropriate in these hydrogeologic settings.

See section 2.5 for further information on hydrogeologic barriers.

Aquifers with thin or absent soil cover - Because fecal contamination sources are assumed to be located at or near the surface, the presence and thickness of soil may be important to attenuating infectious pathogen risk for drinking water wells. Soils are defined herein as unconsolidated material formed in place by natural processes from geologic parent material. Thick soils have enhanced capability to remove pathogens as compared with thin soils. Soil typically has high natural organic matter content (as opposed to material added by septage input) which is relatively efficient at retarding pathogen transport by favoring attachment to soil particles. Where soils are red due to the presence of iron oxides or other favorable soil conditions occur, pathogens may also be efficiently removed. In general, septic tanks are permitted in soils that exhibit variable saturation conditions because such conditions enhance removal of pathogens and other septage contaminants. On the other hand, septic tanks are generally prohibited from continuously saturated soils which do not exhibit such conditions. Where soils are thin or absent, pathogen removal from septage and sewage is minimized and infectious microorganisms are more likely to reach a PWS well tapping an aquifer unprotected by thick soils. However, thick soils can have macropores, root casts, and other openings that mimic the conduits and fractures typical of sensitive aquifers. Where present, macropores can allow direct and efficient transport of fecal contamination through soil, thereby by-passing some of the protective properties of thick soils

Soil formation occurs continuously, but soils may be thin or absent if erosion by wind, water, anthropogenic activity, or glaciers predominates over soil formation processes. In general, humid climates have thicker soils than arid climates. Unglaciated terrain has thicker soil than glaciated terrain. States located in the western US are more likely to be arid, and States located in the northern US are more likely to have had glacial removal of soil. More detailed data on soil presence or thickness are typically available from US Department of Agriculture (USDA) County Soil Surveys (maps and reports) as discussed in Section 1.5.3.

Where soils are thin or absent, the soil capacity to remove or inactivate pathogens is minimal. Assessment source water monitoring may be appropriate for PWS wells located in areas with thin or absent soil (thick soils with soil macropores may also perform poorly).

Because some fecal contamination sources such as septic tanks release fecal indicator microorganisms into the shallow subsurface, these indicators may be insufficiently attenuated when the soil is thin. The variably or completely saturated bedrock may allow passage of indicator microorganisms at greatly differing rates, depending on the bedrock type. Since no fecal indicator type has a greater likelihood of passaging to the wellscreen, any of the three fecal indicator organisms would be appropriate.

Wells previously identified as having been fecally-contaminated - All PWSs must conduct monitoring under the Total Coliform Rule (TCR) and that monitoring may identify fecal contamination of ground water sources. The TCR-approved method identifies many but not all *E. coli* serotypes. For example, the *E. coli* O157:H7 serotype is not identified using TCR-approved methods. Although *E. coli* is used as a fecal indicator organism, some serotypes such as *E. coli* O157:H7 are frank pathogens. In particular, *E. coli* O157:H7 may cause kidney failure in children and the elderly. Wells with a history of *E. coli* contamination, where identified, are more likely to experience additional fecal contamination and can be identified as wells with greater likelihood of infectious pathogen contamination. As part of the sanitary survey, the fecal contamination history of a PWS well would be reviewed. The Sanitary Survey Guidance Manual for Ground Water Systems provides additional discussion about identifying PWS well fecal contamination history.

Wells and systems with a previous history of fecal contamination such as *E. coli* occurrence should be considered to be at high risk from fecal contamination and should be considered for assessment source water monitoring.

The choice of fecal indicator microorganism should be governed by the available data. All indicator microorganism groups (and probably types within each group) are transported at differing rates in the same aquifer materials. If the well has a history of *E. coli* occurrence, then *E. coli* should be selected as the recommended indicator microorganism. It is less likely that wells will have a history of enterococci or coliphage occurrence but if a well has previous occurrence of one of these organisms, then, based on that fecal indicator occurrence history, that fecal indicator should be selected because the aquifer materials appear to permit efficient transport of that indicator to the well.

High population density combined with on-site wastewater treatment systems - Any aquifer may be at risk of fecal contamination if the aquifer’s natural attenuation capabilities are overwhelmed. For example, the high-density fecal contamination discharged into the subsurface by septic tank drainfields and other on-site wastewater treatment systems can pose such a risk. Greater population density combined with restricted areal extent of an aquifer is an especially risky combination because aquifer recharge by septage discharge is significant compared with infiltrating precipitation. Some aquifers, such as barrier island or marine island aquifers, are capable of supplying only limited yield because over-pumping will result in seawater intrusion, permanently damaging the aquifer. Where population density is high and yield is limited, dilution and other natural attenuation processes are also limited, and fecal contamination is more likely. PWS wells located in resort island communities should be targeted for additional monitoring.

Large numbers of visitors to an island, combined with on-site wastewater treatment facilities, are important risk factors that have contributed to aquifer contamination. Localities with large transient populations and on-site wastewater treatment may be considered for assessment source water monitoring to provide additional public health protection.

As discussed in the Sanitary Survey Guidance Manual for Ground Water Systems, land use and sanitary setback distances around PWS wells are important elements of a sanitary survey. It is not the purpose of this document to address land use. However, resort communities with large seasonal population changes are relatively easy to recognize. These locations are likely to have high population density and restricted aquifer yields, and some have on-site wastewater treatment. In such locations, assessment source water monitoring will provide additional public health protection.

Resort communities may be located on, for example, barrier island sand aquifers or karst limestone islands. Because resort island aquifers types are varied, the ground water flow and the subsurface passage of fecal indicator microorganisms is also highly variable. Thus, no fecal indicator is favored and any of the three recommended indicator organisms may be appropriate. However, as discussed previously, sand aquifers may be an aquifer type in which viruses are favored for transport as compared with bacteria and thus coliphage might be a more appropriate fecal indicator organism.

Other Risk Factors – States may have information collected during sanitary surveys, Source Water Assessments, and field visits that indicate a ground water source may be subject to fecal contamination. States may require PWSs with these sources to conduct source assessment monitoring based on this information. Examples of deficiencies or information that may indicate ground water sources are at risk from fecal contamination and therefore indicate that a need to conduct assessment source water monitoring include:

- Well near a source of fecal contamination;
- Well in flood zone;
- Improperly constructed well (e.g., improper surface or subsurface seal);
- Well that does not meet State codes or standards (e.g., depth or setback requirements);
- Well of unknown construction (e.g., no driller’s log or other record of construction); and
- Other non-microbial indicators potential for fecal contamination (e.g., MBAS, high chloride or nitrate levels from baseline, or historical trends).

Due to the great variability among other possible risk factors, no specific fecal indicator organism would be favored and thus any of the three GWR fecal indicator organisms is appropriate.

1.5 Hydrogeologic Data Sources for Assessment Source Water Monitoring Decisions

A number of EPA publications provide detailed discussions of hydrogeologic data sources. An EPA workgroup was convened in 1993 to develop a guidance document on ground water resource assessment. The guidance describes sources of hydrogeologic data and how these data may be used to evaluate aquifer sensitivity (USEPA 1993a). EPA also published the *Ground Water Information Systems Roadmap, A Directory of EPA Systems Containing Ground Water Data* (USEPA 1994a). An EPA Handbook titled *Ground Water and Wellhead Protection* (USEPA 1994b) also summarizes hydrogeologic data sources.

This section augments the discussions contained in these earlier documents, emphasizing desktop analyses. In cases where desktop analysis is not possible because the needed data are not available or do not have sufficient resolution, field investigations may be necessary.

1.5.1 State and Federal Hydrogeologic Investigations

The data sources described in this section are electronic or hard copy reports and/or data produced through previous desktop analyses or field investigations. Such information may have been generated to meet the requirements of SWAPs or through water quality and/or water supply investigations initiated at the local, State, or Federal level.

Existing data for a given PWS well may be used. For example, if an existing report or appropriate scale map indicates whether or not a PWS well is screened in a sensitive (i.e., karst, fractured bedrock, or gravel) aquifer, then that information can be used to satisfy the HSA requirement. Generally, spatial data at the scale of 1:100,000 or larger (e.g., 1:24,000) are sufficiently detailed for most purposes [Note: large scale maps provide detailed information of small geographic areas.]

1.5.1.1 Wellhead Protection and Source Water Assessment Studies

The SDWA, as amended in 1986, created the Wellhead Protection Program (WHPP). Each State is required to adopt a program to protect wellhead areas within its jurisdiction from contaminants that may have adverse health effects and to submit the program plan to the EPA Administrator. Currently, 49 States and two territories have WHPPs in place. In their WHPPs, States address all program elements including how to delineate wellhead protection areas (WHPAs) and how to identify and inventory all potential sources of contamination.

Section 1453 of the 1996 SDWA Amendments required all States to establish SWAPs and to submit plans to EPA for approval by February 6, 1999. These SWAPs address both surface water and ground water protection, and their SWAP plans detail how States will: (1)

delineate source water protection areas; (2) inventory significant contaminants in these areas; and (3) determine the susceptibility of each public water supply to contamination. States may use any available information to carry out the SWAP, including data generated through the WHPP. After plan approval, the States must have completed susceptibility determinations for all PWSs by November 6, 2001, unless the State was granted an 18-month extension until May 6, 2003.

EPA encourages States to build upon previous SWAP or WHPP efforts to determine hydrogeologic sensitivity. A review of selected, approved State SWAP plans across EPA regions indicates that many States intend to evaluate hydrogeologic information that may enable them to determine a PWS well's aquifer type.

At least one State addresses the three sensitive aquifer types (i.e., karst, fractured bedrock, and gravel) and the presence of a hydrogeologic barrier as part of a susceptibility determination process in its approved SWAP plan (WIDNR 1999). Other approaches to fulfilling SWAP requirements are also likely to result in data that will be useful for source water assessment. Case studies # 2 and # 4, presented in sections 3.2.2 and 3.3.2, respectively, illustrate just two ways in which data can be extracted from SWAP investigations.

1.5.1.2 State Geologic Survey, USGS, and Other Hydrogeologic Investigations

Many State geologic surveys and/or agencies of natural resources have significant experience studying local and regional aquifer systems and investigating ground water quality and quantity issues. Although many of these studies may have directly supported, or continue to support, SWAP or WHPP work, many more studies have been conducted independently of these efforts. In addition to State geologic surveys, the United States Geological Survey (USGS) has district offices that perform similar work in each State, sometimes in cooperation with State agencies. Universities, local governments, and non-governmental organizations also conduct pertinent hydrogeologic research.

1.5.2 Hydrogeologic and Geologic Maps

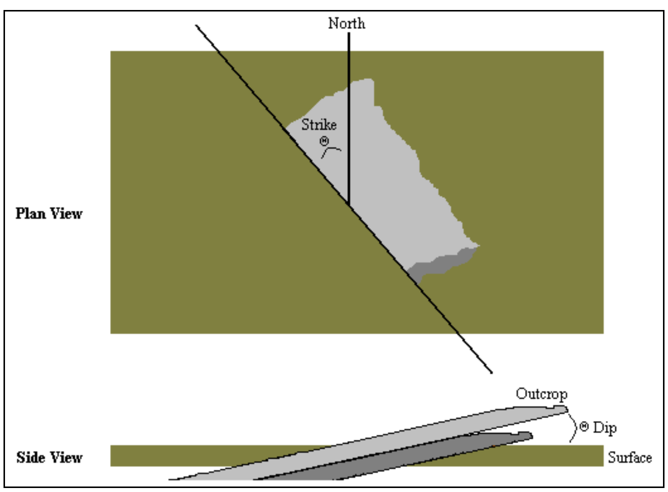
Hydrogeologic or aquifer maps generally show the location, spatial extent, and depth of aquifers in a region. Such maps typically include information on aquifer type as well. Hydrogeologic maps will often be the most direct means of evaluating important risk factors such as aquifer type.

Geologic maps may depict a region's surficial geology, which would include the locations and extent of distinct unconsolidated deposits and bedrock units exposed at the earth's surface, or, alternatively, the bedrock geology of an area. Surficial geologic maps are available for many areas from the USGS and often include a key to interpret the results of various test holes shown on the map. Using geologic maps is a less direct means of identifying aquifer type than using hydrogeologic maps. But analytical techniques such as projection (described below) and information such as well depth can help in determining aquifer type.

The availability of hydrogeologic maps at an appropriate scale varies among States and among regions. The following sources may be useful to States in obtaining appropriate maps for use in preparing HSAs. As part of its Regional Aquifer-System Analysis (RASA) program, the USGS has produced a large variety of hydrogeologic maps at various scales. Some of these maps are at scales that may be useful for an HSA (Sun et al. 1997). The RASA program completed studies of 25 major US aquifer systems in 1995. *The Ground Water Atlas of the United States* was developed as part of the RASA program and provides small-scale (i.e., numerically large, less detailed coverage of large geographic areas) hydrogeologic data for the country both as a printed atlas and as a digital dataset (available on the Internet at: http://capp.water.usgs.gov/gwa/). The printed atlas has 13 individual chapters that cover specific US regions. *The Ground Water Atlas* data, however, are compiled at scales that may not be suitable for public water system-specific HSAs (e.g., at 1:5,000,000 and 1:2,500,000 scales).

In areas where hydrogeologic maps are not available, a geologic map along with the projection method may be used to determine the aquifer type for a well of a given depth. Projection is a structural geologic technique which can be used to determine aquifer depth, or the depth of any local geologic unit at a well, using the strike and dip of the aquifer as measured at nearby outcrops. Typically, bedding (layering) can be described in terms of its strike and dip. Bedding occurs (but may be indistinct) in some sedimentary rocks, in metamorphosed sedimentary rocks (metasediments), and in some igneous rocks such as volcanic flows (e.g., basalts). Outcrops of the bedrock are shown on many geologic maps along with the values of the strike and dip of the bedding. The strike is the compass direction or azimuth of the line formed by the intersection of the bed with its horizontal (planar) surface. The dip is the angle in degrees between the bedding and a horizontal surface, measured at right angle to the strike (Exhibit 1.4). If the bedrock is a known aquifer, the depth to that aquifer can be determined by projecting the dip over the distance to the well location. Using simple trigonometry, the depth to the aquifer is then equal to the tangent of the angle multiplied by the distance. This method can be used in areas of simple geology.

Exhibit 1.4 Strike and Dip



More detailed hydrogeologic and geologic maps are available from a variety of public and private entities. The USGS, as well as State geologic surveys or natural resources agencies, are the most prolific sources. However, coverage is highly variable from State to State. The National Research Council (NRC) estimated in 1988 that less than 20 percent of the United States has been geologically mapped at a scale of 1:24,000 or larger (NRC 1993).

In response to this situation, Congress enacted the National Geologic Mapping Act of 1992. This act established the National Cooperative Geologic Mapping Program (NCGMP) to implement expanded geologic mapping efforts through a consortium of geologic mappers. As part of this program, the USGS conducts Federal mapping projects through its FEDMAP program; STATEMAP, run by State geological surveys, is a matching-funds grant program; and universities participate in another matching-funds program - EDMAP. The USGS coordinates the NCGMP, which has a long term goal of producing 1:24,000-scale geologic maps for high-priority areas of the States and national coverage at the 1:100,000-scale.

The NCGMP also maintains an exceptionally useful database for locating existing geologic maps produced by a wide variety of entities, and it includes mapping currently in progress through the consortium and is searchable by location, scale, and other parameters. The database, as well as general information on the program, is available on the Internet at http://ngmdb.usgs.gov/. A geologic map index is also available for many States showing boundaries for compiled map projects and references.

1.5.3 Soil Maps

The Natural Resources Conservation Service (NRCS), a division of the United States Department of Agriculture (USDA), is responsible for soil mapping in the United States. Soil survey maps have been completed for most of the United States, as of December 2002 (NRCS-NSSC 2002). The mapping is done using aerial photographs at scales that depend upon needs, but scales of 1:24,000, 1:15,840, and 1:12,000 are common. Soil maps are published as Soil Survey Reports, usually on a county by county basis.

Although they do not provide direct information regarding aquifer type, careful interpretation of county soil surveys can yield useful information on other risk factors such as presence or absence of thick protective soil. Soil maps are accompanied by detailed descriptions of the mapped soil series which indicate the earth materials from which they were derived (i.e., the "parent materials").

In some cases, the parent materials may be the underlying bedrock. Soils formed in place by the weathering of bedrock are called residual soils. A soil may also form in sediments transported to the site (e.g., by stream or glacial deposition). Therefore, even though the soil series descriptions always indicate a soil's parent material, only in the case of residual soils will that information directly indicate the type of underlying bedrock, and possibly the aquifer type for the PWS of interest. Nonetheless, the underlying bedrock type may be noted in the soil series description even if it is not the parent material. For example, in the *Soil Survey for Essex County, Massachusetts, Southern Part*, the Chatfield series profile description notes that the parent material is glacial till, but granitic bedrock is found 34" below the land surface. Soil series profile descriptions are based on a profile from the survey area that is considered typical of the given series (USDA-SCS 1984).

The availability of soil survey maps and suggestions for how to obtain maps not available on the Internet can be checked on the Internet at http://soils.usda.gov/survey/. Maps in the "published" category can be considered up-to-date. The "initial mapping complete" category refers to maps that can be obtained either on CD from NRCS or as a hard copy, per request by the State. Similarly, areas listed as "Update Field Work Complete" may be available on the Internet, and areas listed as "Update Field Work in Progress" are likely to have maps available upon request to the NRCS. Maps in the categories of "Maintenance Needed" or "Maintenance" are likely to require some minor changes (e.g., due to regional floods changing the character of a floodplain, as occurred with the Mississippi River floodplain in 1993). Areas on this map that are considered "non-project" by the NRCS may still have soil survey maps that can be obtained from the Federal agency that has responsibility for the land in that area. Alternatively, local NRCS field offices may also have data for "non-project" sites. Areas that are listed as "initial mapping in progress" (only 5 percent of the United States) do not have soil survey maps currently available.

Digital soil survey data are also available. The Soil Survey Geographic (SSURGO) Data Base is comprised of digitized county-level soil survey maps (compiled at scales ranging from 1:12,000 to 1:63,360), and are of much higher resolution than STATSGO data. Therefore, SSURGO data are more suitable for performing HSAs. Furthermore, SSURGO data are designed for use with GIS because they are linked to a Map Unit Interpretations Record (MUIR) attribute data base. SSURGO data are not yet available for every county and area, however the database continues to grow. States can easily check to see if a soil survey is available for a particular county by accessing the SSURGO list. All soil surveys on this list can be obtained either on the Internet, or are available on CD by requesting the information from the NRCS. The SSURGO data available on the Internet are updated monthly.

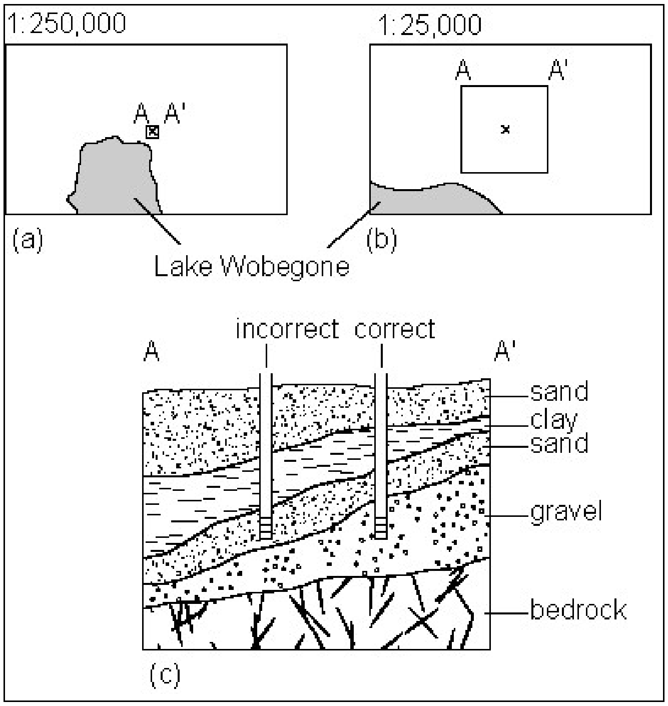
1.5.4 Topographic Data

Well coordinates, depth to the screened interval of a well, and topographic maps (described below) can be used to determine whether or not a well is drawing water from a given aquifer. Imprecise plotting of a well's location could lead to an erroneous assessment of the aquifer type from which the well is drawing water (and thus possibly an incorrect evaluation of whether or not the well is drawing from a sensitive aquifer).

Accurate determinations of well locations are critical for making sensitivity determinations using a desktop analysis; thus, it is important to use large scale topographic maps (e.g., 1:24,000 topographic quadrangles) for plotting the well's location (see Exhibit 1.5). In the absence of a detailed topographic map (e.g., 1:24,000), a base map of comparable scale is needed to accurately locate the well. Such a map might be available from the local community (e.g., Assessor's Office, Engineering Department, Department of Public Works, Water Board, Board of Health, Planning Board, and Conservation Commission) or from State, Federal, or regional natural resource agencies and planning departments.

Accurate well coordinates may be sought first from the PWS. Well registration information collected by Federal, State, and local regulatory programs also usually include coordinates. They may also be available from the well drilling company records. If necessary, well coordinates can also be obtained in the field using Global Positioning System (GPS) technology. The City of Tallahassee (1996) has described the process of locating PWS wells using GPS receivers and discussed the important issue of receiver accuracy.

Exhibit 1.5 The Importance of Map Scale for Determining Aquifer Type



In Exhibit 1.5, X indicates the location of a well with known areal coordinates and depth. On the smaller scale map, (a), precise plotting of the well's location is impossible. The larger scale map, (b), shows the location of X with much greater precision. Cross-section (c) shows a correct identification, based on map (b), of the well's aquifer as gravel (a sensitive aquifer) and an incorrect identification, based on map (a), of the well's aquifer as sand (a non-sensitive aquifer).

Topography can be represented in two dimensions with contours, continuous lines that join points of equal value (equal elevation in this case). The contour interval, which is the change in elevation between each successive contour line (e.g., 20 feet), is chosen depending upon the scale of the map and the topographic relief. The USGS and the Defense Mapping Agency (DMA) have produced most of the topographic maps for the United States (NRC 1993). The USGS produces maps at a variety of scales, but the most common scales for topographic maps are 1:24,000/1:25,000, 1:100,000, and 1:250,000. The 1:250,000 scale maps are available for the entire United States. The much more detailed topographic quadrangles (1:24,000 or 1:25,000) are available for most of the country. Index maps showing available topographic maps for each State are provided by the USGS without charge. Each 1:24,000 topographic map covers approximately 58 square miles, where 1 inch corresponds to 2,000 feet.

Digital topographic data for the United States are also available from the USGS as Digital Line Graphs (DLGs) and Digital Elevation Models (DEMs). DLGs are vector data files that represent linear and areal features commonly found on topographic maps, including contour lines. DEMs are data files that store point elevations spaced at regular intervals in a matrix. Detailed DEMs have 10- and 30- meter resolutions. Because national coverage is incomplete for both DLGs and DEMs, and State-wide coverage varies considerably by State, the remainder of this section will focus on paper topographic quadrangles.

Topographic maps are based on aerial photos, and skilled topographic map interpretation may reveal landform features such as sinkholes and "losing streams" (in this context, a stream that disappears and loses its water to ground water via an underground route). Such features are indicative of the underlying bedrock type and/or structure (in cases where the structure controls the topography). Discontinuous drainage networks are also revealed on detailed topographic maps and indicate a karst environment. Drainage may follow the underground joint pattern in the rock, which is expressed on the topographic map. Contour lines representing elevation may also reveal distinct features of the local bedrock structure such as folds and faults; such structures are almost invariably associated with fracturing. The topographic map will also show the orientations of folds and/or faults that have a surface expression, helping to establish the orientations of regional fracture networks.

The surficial or geomorphic features associated with a particular soil type may be represented on a topographic map. Deposits likely to consist of coarse gravel can be readily identified on a topographic map by their surface expression. The geometries or drainage patterns of streams can provide clues to the underlying geology. A dendritic drainage pattern will most likely be found in horizontal sedimentary rocks or massive igneous rocks, but can also be seen in folded or complex metamorphic terrain. Trellised and rectangular drainage patterns indicate faulted and jointed rock. Centripetal patterns, with or without trellised drainage, can indicate the presence of sinkholes. In areas covered by overburden, the lack of surface streams is an indicator that well-draining granular soils underlie the area.

1.5.5 Stereoscopic Aerial Photography

Aerial photographs taken with approximately 30 percent overlap allow three dimensional imaging of land surface features with the aid of stereoscopes. In regions with limited geologic or topographic data, stereoscopic air photos may help locate wells. In most cases, however, such photos will be most useful for determining aquifer types when used in conjunction with other data sources. For example, if low resolution geologic maps or well log data indicate that a given PWS well may be screened in a karst aquifer, stereoscopic aerial photos could be used to determine the presence or absence of sinkholes and/or other characteristic karst landform features.

Aerial photographs are available from several entities within the USDA and from the USGS. The NRCS and the Forest Service, both under the USDA, have extensive U.S. coverage at scales appropriate for HSAs. As noted above, the NRCS uses high resolution aerial photography to compile their county level soil surveys at scales ranging from 1:12,000 to 1:63,360 (see section 1.5.3). The USDA Aerial Photography Field Office, Farm Service Agency acts as the clearinghouse for all USDA aerial imagery, archiving over 10,000,000 images dating to 1955. USDA aerial photo coverage, availability, and ordering information are available through their website at: http://www.apfo.usda.gov/.

The USGS National Mapping Division administers the National Aerial Photography Program (NAPP). The NAPP coordinates the collection of cloud-free coverage of the conterminous United States and Hawaii at a uniform scale (approximately 1:40,000) about every five years. NAPP photographs are available in black-and-white, and in many cases, color infrared. The imagery is available from the USGS's Earth Resources Observation Systems (EROS) data center (http://edc.usgs.gov/) or Earth Science Information Centers (ESIC). NAPP photos are also available from the USDA Aerial Photography Field Office, Farm Service Agency (see link above).

1.5.6 Other Data Sources for Desktop Analyses

Well registration information and well logs collected by local, State, and Federal regulatory programs may be very useful for determining aquifer type. Well registrations usually indicate well locations and information necessary to conduct an HSA. A sufficiently detailed driller's log for a PWS well could itself, or in combination with other data sources, adequately characterize the subsurface stratigraphy and aquifer type. For example, based upon a regional bedrock geology map that is of moderately low resolution (e.g., 1:700,000), a State may identify that a PWS well is located in an area underlain primarily by limestone. The State may review the driller's log (if available) to confirm that, in fact, the well is screened in a limestone aquifer. Certain States such as New Jersey and New Hampshire require drillers to file a log for each well with the appropriate State agency, such as a water well board or the State environmental protection agency.

A driller's log typically records changes in lithology with depth, although local terminology may be used and may need deciphering. For example, in much of the United States the term "artesian well" is used by drillers as a lay term to indicate a producing bedrock well. This contrasts with the hydrogeologist's definition - a confined aquifer where the water in a well rises above the top of the aquifer, sometimes flowing to the land surface. Another example is the use of the term "hardpan" by drillers to describe what may be a dense glacial till, a cemented soil, or a hard clay. A driller's log may also include information on the drilling method employed, which may give clues to the type of materials the drillers encountered.

Additional desktop sources include consultant reports and database searches for property site assessments conducted by private search companies. These searches of Federal, State, and local agency databases are conducted as part of due diligence investigations for property site assessments and are usually in accordance with the standards of the American Society for Testing and Materials (ASTM). These database searches include a description of the bedrock and surficial geology, a well inventory, and usually aerial photo coverage for the area in question. The well inventory summarizes well locations, construction, soil and bedrock type, water quality, and other pertinent data.

2. Sensitive Hydrogeologic Environments

2.1 Aquifer Sensitivity

Aquifers in which pathogens can move quickly to public supply wells, allowing little filtration or time for inactivation, are considered sensitive. These aquifers generally have rapid ground water flow velocities (that increase in the vicinity of pumping wells) and short, direct flow paths. Three aquifer types are most likely to have these properties: karst, fractured bedrock, and gravel aquifers. These aquifer types are described below in sections 2.2, 2.3, and 2.4, respectively. Exhibit 2.1 provides examples of sensitive aquifers and their common names as well as examples and sources of USGS maps of aquifer types. Exhibit 2.2 provides examples of hydrogeologic barriers and their properties.

Aquifer type, which is usually well-correlated with lithology, is important and should be identified to determine if a specific aquifer is sensitive. This document includes all limestone aquifers, igneous and metamorphic aquifers, and gravel aquifers as sensitive. This is because limestone is the lithology most likely to be karst; igneous and metamorphic aquifers are likely to be highly fractured; and gravel aquifers are likely to have direct flow paths and rapid ground water velocities due to the shape and large size of their pores. Pumping wells increase the natural flow velocities in a sensitive aquifer to a greater degree than they would in a fine-grained, unconsolidated aquifer, for example.

States may designate additional aquifer types as sensitive if they believe it is necessary to do so to protect public health (e.g., a State may designate sand and gravel aquifers as sensitive). This guidance will not cover other potentially sensitive aquifers because States have the flexibility to set their own criteria regarding other aquifer types that may or may not be sensitive.

The following sections - 2.2 through 2.4 - describe the rationale for including or identifying karst, fractured bedrock, and gravel aquifers as sensitive. Brief summaries of bacterial contamination research in each of these aquifer types are included, as well as summaries of known disease outbreaks resulting from such contamination. Because karst aquifers make up such a large proportion (40 percent) of all productive aquifers in the United States (USGS, 2002), section 2.2 describes the surface and subsurface hydrologic characteristics of karst regions, and the potential for bacterial contamination of karst aquifers, in some detail. Nevertheless, fractured bedrock and gravel aquifers are considered equally sensitive, and when encountered, are worthy of the same time and consideration as karst aquifers.

Exhibit 2.1 Example Sensitive Aquifers

Aquifer Type	Common Name	Example	USGS Classification and Map of Aquifer Types in US
Carbonate	Karst	Limestone	http://capp.water.usgs.gov/aquiferBasics/carbrock.html#list
Igneous	Fractured bedrock	Granite	http://capp.water.usgs.gov/aquiferBasics/volcan.html
Metamorphic	Fractured bedrock	Gneiss	http://capp.water.usgs.gov/aquiferBasics/volcan.html
Unconsolidated	Gravel	Glacial outburst deposits	http://capp.water.usgs.gov/aquiferBasics/uncon.html

Exhibit 2.2 Examples of Hydrogeologic Barriers and their Properties

Barrier	Porosity[1] (%)	Permeability Range[1] (cm/s)	Specific Yield (%)	Hydraulic Conductivity[6] (cm/s)
Clay	45-55	(10^{-6} - 10^{-8})	(1 - 20)[2]	1.4×10^{-6} to 1.4×10^{-9}
Glacial Till	45-55	(10^{-6} - 10^{-8})	(5 - 20)[3]	-
Shale		(10^{-6} - 10^{-8})	(0.5 - 5)[4]	-
Siltstone		(10^{-6} - 10^{-8})	(1 - 35)[5]	1.4×10^{-6} to 9.4×10^{-10}

[1]Brown et al., 1983

[2]Depends on source. Heath, 1983; Morris and Johnson, 1967, as compiled by McWhorter and Sunada, 1977; Sevee, 1991; Devinny et al, 1990

[3]Devinny et al, 1990

[4]Sevee, 1991

[5]Depends on source. Morris and Johnson, 1967, as compiled by McWhorter and Sunada; Sevee, 1991; Devinny et al, 1991

[6]Compiled from Morris and Johnson, 1967, by Barton et al, 1985

2.2 Karst Aquifers

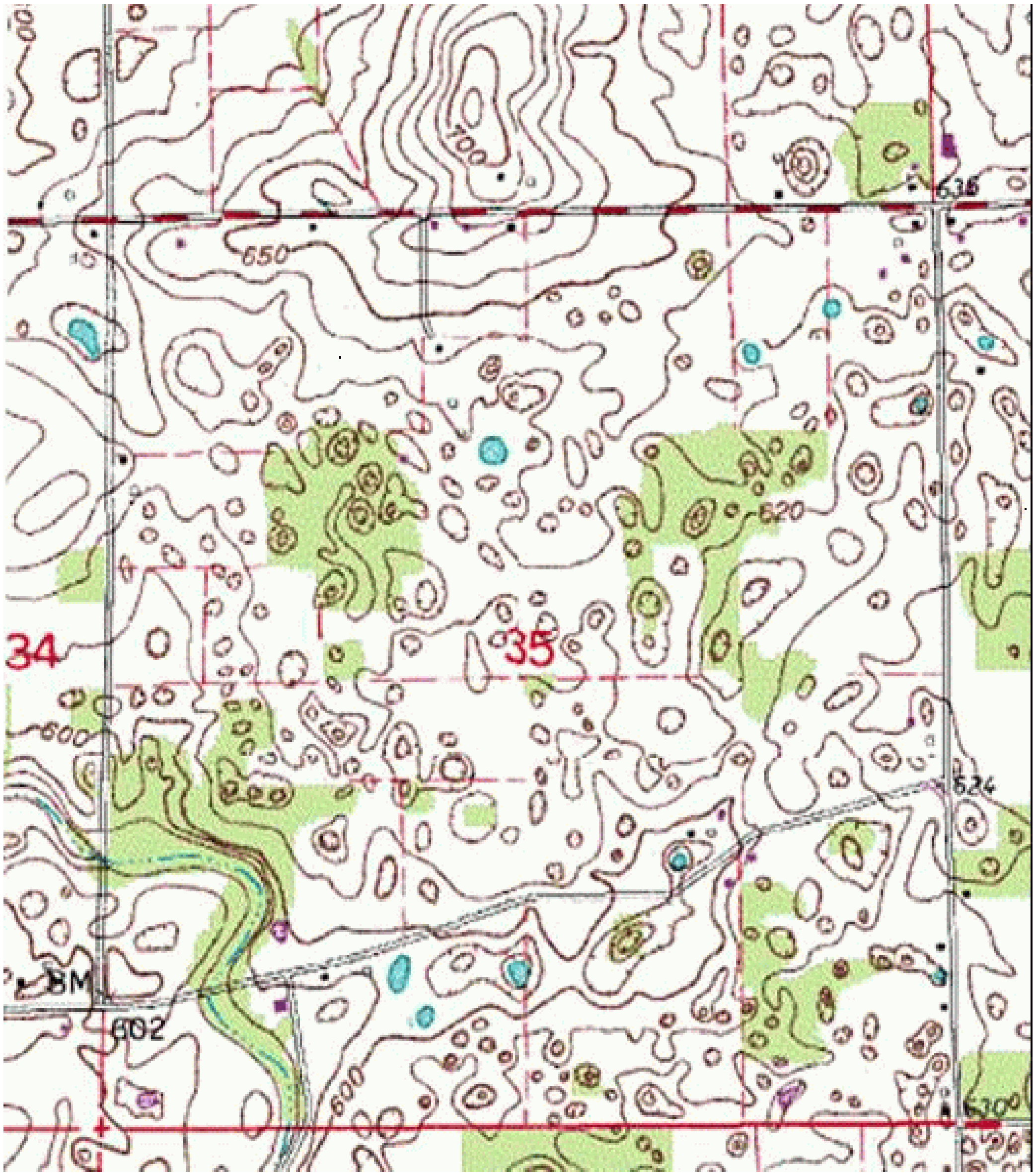
Karst is defined as a type of geologic terrain within which flowing ground water has dissolved significant portions of the area's soluble (usually carbonate) rocks (Fetter 2001). Where karst regions occur, infiltrating precipitation and ground water create a permeability structure characterized by numerous and often large, interconnected conduits. Through time, these conduits continue to enlarge, creating unique surface and subsurface drainage networks and characteristic surface landforms. Ground water velocities are usually rapid and flow paths are very direct in karst environments, especially in the vicinity of pumping wells. All limestone aquifers are designated as sensitive aquifers in this document, due to the likelihood that they are karst environments. Microbial pathogens released into karst aquifers that intersect the ground surface from subsurface sources such as septic systems or surface sources such as livestock feedlots are likely to reach drinking water consumers in an infective state. For example, the Walkerton, Ontario *E. coli* outbreak in May, 2000 is believed to have been caused by fecal pollution of a karst aquifer system (Worthington et al. 2001, 2002). Two outbreaks in Braun Station, TX (D'Antonio et al., 1985), and outbreaks in Georgetown, TX (Hejkal et al., 1982) and in Brushy Creek, TX (Bergmire-Sweat et al., 1999; Lee et al., 2001) all resulted from contaminated wells located in the Edwards Aquifer, a sensitive karst limestone aquifer.

Similarly, outbreaks in South Bass Island, OH (Ohio EPA, 2005; USCDC, 2005; Fong et al., 2007; O'Riley et al., 2007) and Walkerton, Ontario (Golder Associates, 2000; Health Canada, 2000; Hurley and Hurley, 2004) resulted from contaminated wells located in the Upper Silurian Bass Island Formation, a sensitive karst limestone aquifer.

Karst features are most commonly found in limestone, but marble, dolomite, evaporites (e.g., gypsum), and other soluble rocks may also have karst features. Calcium magnesium carbonate is less soluble than calcium carbonate, which is why limestone (composed primarily of calcium carbonate) is more likely to be karstic than dolomite (composed of calcium magnesium carbonate) (Freeze and Cherry 1979). Solution-enlarged fractures and conduits - typical karst features - tend to provide the dominant ground water flow paths in limestone aquifers. For the purposes of this guidance manual, it is assumed that all limestones are karstic and are therefore sensitive. Other potentially soluble rocks such as dolomite may also be recognized as sensitive aquifers, especially if the dissolution openings are enhanced by fracturing (e.g., Door Peninsula, WI). States may choose to consider all carbonate (potentially soluble) aquifers, and other aquifers formed by soluble rocks (e.g., evaporite aquifers in the Las Vegas Valley; see USGS Circular 1170, 1998), as possibly karstic and designate them sensitive.

Karst regions are typically characterized by the following: underground drainage networks with solution openings that range in size from enlarged fractures to large caves; closed surface depressions, known as sinkholes, where the dissolution of the underlying bedrock has caused the collapse of overlying rock and sediment; and discontinuous surface water drainage networks that are related to the unique subsurface hydrology (Winter et al. 1998). In areas such as Orange County, Indiana, there are so many sinkholes (over 1000 per square mile, Exhibit 2.3) that they coalesce into compound sinkholes (Thornbury 1954). In other mature karst landscapes, characterized by relatively pure limestone in areas of high precipitation, caves and caverns are formed in the subsurface. Conduits in carbonates and gypsum can be quite large with some exceeding 100 feet in diameter (i.e., caves) and several miles in length. Mammoth Cave, Kentucky has a mapped length of more than 340 miles of interconnected conduits distributed over five horizontal levels. Ground water velocities have been measured there at more than 1,000 feet per hour (USEPA 1997).

Exhibit 2.3 Map of Sinkholes (closed circular contours with tick marks) in Orleans, Indiana



Indeed, it is the rapid ground water velocities in karst aquifers that necessitate their characterization as sensitive aquifers. In the karst region of Slovenia, bacteriophage injected into a karst aquifer reportedly traveled approximately 24 miles in less than 4 months (Bricelj 1999). Using conservative ground water tracers, scientists have measured ground water velocities in karst aquifers to be as high as approximately 0.3 miles per hour (USEPA 1997). In Florida, ground water velocities surrounding a well have been measured at several hundred feet per hour (USEPA 1997). In a confined karst aquifer in Germany which was breached by monitoring wells, ground water traveled approximately 650 feet in less than 4 days (Orth et al. 1997). In the Edwards Aquifer, Texas, Slade et al. (1986) reported that dye traveled 200 feet in 10 minutes. This data all indicates that ground water flows extremely rapidly through karst aquifers.

Well-developed karst systems may have underground streams because of the large size of interconnected openings in the rock. Underground streams can have flow rates as great as those of surface streams. It is also not unusual in karst terrains for surface streams of considerable size to disappear into solution cavities (swallow holes) intersecting a streambed, creating a discontinuous surface drainage system. These same streams may reappear at the surface at other locations (Winter et al. 1998). Seeps and springs are thus common in karst regions.

Sinkholes in karst regions can play a particularly devastating role in the microbial contamination of ground water supplies. For example, sewage treatment lagoons have been known to leak and eventually collapse over sinkholes. This phenomenon has been documented in West Plain, Missouri in 1978 (Craun 1984); in Lewiston, Minnesota in 1991; and in Altura, Minnesota in 1974 and 1976 (Jannik et al. 1991). In Missouri, 759 illnesses resulted from the contamination of domestic wells due to this 1978 sinkhole collapse (Craun 1984).

Even in the absence of sinkhole collapse, the potential for rapid infiltration of fecal contamination through overlying soils into karst aquifers is great when karst aquifers intersect the ground surface. Residual soils, formed by bedrock dissolution, are characteristic of well-developed karst regions. These soils are typically clay-rich, but can have great variation in thickness and hydraulic conductivity (the capability to transmit water). Soil macropores transmit water rapidly, and are caused by channels formed by decayed roots, insect and animal burrows, dessication cracks, soil failure surfaces, and soil piping (USEPA 1997). Rapid flow in the overlying soil may also occur via vertical fissures, even when there is substantial residual soil cover (Smart and Frederich 1986, cited in USEPA 1997). Where the mantle of glacial till or outwash deposits is thin, infiltration velocities may also be high (Crowther 1989, cited in USEPA 1997).

The actual transport of fecal bacteria within karst aquifers has been studied at a variety of localities (Malard et al. 1994; Orth et al. 1997; Tranter et al. 1997; Gunn et al. 1991). Malard et al. (1994) suggested that both fractures (discussed in section 2.3) and karstification contribute to rapid bacterial transport in limestone. For this reason, Malard et al. (1994) consider the risk of bacterial contamination greater in limestone than in any other type of aquifer.

It is important to note that concentrations of bacteria within karst environments often vary significantly with rainfall. Personne et al. (1998) found that high aquifer water levels, induced by high rainfall, correlated with high bacteria levels in the aquifer. The water level in one Edwards Aquifer well (582 feet deep with a water table 240 feet deep) began rising within 1 hour after a rainfall event (Slade et al. 1986). Mahler et al. (2000) studied fecal coliform and enterococci bacteria near a wastewater irrigation site, and found the presence of bacteria in ground water directly followed rainfall events. Mahler’s data suggests that small sampling intervals of 3 to 4 hours are necessary to describe the breakthrough of bacteria at a monitoring well screened in a karst aquifer.

The potential for rapid transport of bacteria and viruses through karst aquifers necessitates that they be monitored carefully for contamination. Bacteria can rapidly percolate into the unsaturated zone of karst aquifers, as well as be farther transported to the saturated zone during periods of intensive rainfall. In fact, Malard et al. (1994) found high occurrence rates for bacteria in a karst aquifer as long as a year after surface pollution had essentially ceased. This data demonstrates that sensitive aquifers can be contaminated even when surface pollution sources are difficult to identify. Furthermore, research shows that surface water and ground water drainage divides generally do not coincide in karst regions due to complex patterns of surface water and ground water flow. For example, a stream may disappear in one surface water basin and reappear in another basin. This situation makes it even more difficult to successfully inventory sources of fecal contamination in the recharge area of a karst well (Winter et al. 1998). Such situations are part of the motivation behind the GWR's focus on monitoring sensitive aquifers, rather than merely looking for potential sources of bacterial contamination. In summary, bacterial contamination of karst aquifers is both fairly likely and highly unpredictable, although correlations with rainfall events are common.

2.3 Fractured Bedrock Aquifers

This document considers all igneous and metamorphic aquifers to be fractured bedrock aquifers and designated sensitive. They are considered hydrogeologically sensitive due to the rapid velocities and direct flow paths through the fractures. Under the influence of pumping, already naturally high flow velocities increase to even higher rates in fractured rock aquifers. In general, fractures have a role in ground water movement through any consolidated aquifer, however fractured bedrock aquifers are those in which fractures provide the dominant flow paths. Other aquifer types that may also be fractured (e.g., sandstone aquifers) are not considered sensitive in this document. Nevertheless, States may choose to investigate the degree to which these other aquifers are fractured, and decide if these aquifers should be monitored for bacterial contamination.

Any solid block of igneous or metamorphic rock (i.e., the matrix) that is surrounded by fractures is considered essentially impermeable (Domenico and Schwartz 1990). Thus, all flow is forced to take place within the fractures. A detailed understanding of flow in a fractured bedrock aquifer requires knowledge of fracture widths, orientations, the degree to which individual fractures are mineral-filled, and the degree of fracture interconnection and spacing. Most fracture widths are smaller than one millimeter (mm), and a fracture's capability to transmit ground water (i.e., hydraulic conductivity) is roughly proportional to the cube of the fracture width (NRC 1990). Thus, small changes in fracture width result in very large changes in hydraulic conductivity. For example, a 1 mm fracture can transmit 1000 times more water than a 0.1 mm fracture, provided that other factors are constant (e.g., hydraulic gradient).

Freeze and Cherry (1979) report void space as high as 10 percent of total volume in igneous and metamorphic rock. Other data presented in Freeze and Cherry (1979) suggest that the first 200 feet beneath the ground surface produces the highest water yields to wells because fractures at shallow depths are wider, more numerous, and more interconnected. Nevertheless, municipalities sometimes derive high volumes of water from wells located in fault zones that extend to depths measured in miles.

EPA (1991a) discusses several basic differences between fractured bedrock aquifers and unconsolidated, granular aquifers (e.g., sand aquifers). In unconsolidated, granular aquifers, flow tends to be slow, laminar, and predictable using Darcy's Law. Such aquifers can more easily be assumed to be homogeneous and isotropic. In contrast, flow through fractured bedrock aquifers may be fast (most commonly 3 to 330 ft/yr; and sometimes over 3,000 ft/yr; USEPA 1987a), and Darcy's Law will often not apply. Flow takes place through fractures rather than through pores between individual grains. Furthermore, fractured bedrock aquifers tend to be more heterogenous (i.e., flow properties vary with location in the aquifer) and anisotropic (i.e., flow properties vary with the direction of flow) than granular aquifers. EPA (1991a) provides more detailed descriptions of each these properties.

Tracer tests have been used in several studies to estimate ground water flow rates in fractured bedrock. Malard et al. (1994) report that dye traveled approximately 140 feet in a fractured bedrock aquifer in 2 hours. Becker et al. (1998) report that water traveled approximately 118 feet in about 30 minutes. Ground water velocities in fractured bedrock aquifers are comparable to velocities in karst aquifers. Thus, fractured bedrock aquifers are vulnerable to contamination by waterborne pathogens.

As with other sensitive aquifers, the rapid ground water velocities in fractured bedrock aquifers provide a means by which pathogenic bacteria or viruses can travel quickly from contaminant sources to PWS wells. It is important to note that the pumping of a PWS well causes a greater increase in ground water velocities in fractured bedrock than it would in a non-sensitive aquifer. Recent cases of waterborne disease outbreaks due to contamination of wells screened in fractured bedrock aquifers occurred in Couer d'Alene, ID (Rice et al., 1999), Big Horn Lodge, WY (Anderson et al., 2003; Gelting et al., 2005) and northern Arizona (Anderson et al., 2003; Gelting et al., 2005) .

2.4 Gravel Aquifers

Gravel aquifers, as defined here, are unconsolidated water-bearing deposits of well-sorted pebbles, cobbles, and boulders. Gravel aquifers consist primarily of coarse grains larger than approximately 4 mm or approximately 0.16 inches in diameter, although they may have minor amounts of smaller diameter material as well. Gravel aquifers are often limited in area and are generally produced by high energy events such as catastrophic glacial outburst floods or flash-floods at the periphery of mountainous terrain. They can also sometimes be found at fault-basin boundaries or in glacio-fluvial deposits such as crevasse fillings, eskers, kame terraces, and outwash/valley trains. Typically, these are small, relatively localized aquifers.

Gravel aquifers are not particularly numerous, as compared with sand and gravel aquifers, karst aquifers, and fractured rock aquifers. Very few PWSs use this type of aquifer and reports of outbreaks are correspondingly limited. No data are available that specifically implicates a gravel aquifer as a contributing factor in a published waterborne disease outbreak. Nevertheless, ground water velocities in gravel aquifers can be quite rapid due to the aquifers' large interconnected pore spaces, offering little resistance to flow. Such velocities increase significantly under the influence of pumping wells. Gravel aquifers thus have the potential to become contaminated with microbial pathogens and are therefore designated as sensitive.

The following paragraphs discuss the formation of gravel aquifers due to catastrophic glacial outburst floods. This information may be useful to States in helping to identify potential locations of sensitive gravel aquifers. The discussion below focuses on the western United States. Additional information on possible locations of sensitive gravel aquifers throughout the country can be found in Chapter 3.

Repeated catastrophic floods, resulting from the breaching of large ice-dammed lakes during glacial periods that ended about 12,000 years ago, are believed to be responsible for the formation of the larger pebble, cobble, and boulder aquifers that are more widely distributed in the western United States (Bretz 1925; Baker et al., 1987). Glacial Lake Missoula, one of the largest glacial lakes of the Wisconsin Glaciation (the most recent glacial period), was estimated to have a maximum water depth at the ice dam of about 2,100 feet (Pardee 1942, cited in Baker et al. 1987). The area inundated by Missoula flooding as a result of the breaking of the ice dam is hypothesized to include large parts of western Washington, Idaho, and Oregon (O'Connor and Baker 1992).

The Missoula floods exhibited exceptional sediment transport capability as evidenced by the size of boulders entrained by the flood waters. Baker et al. (1987) conducted field measurements and performed calculations that estimate the peak Missoula flood discharges at three or four orders of magnitude greater than the modern flood discharges of major rivers such as the Amazon or the Mississippi. The floods produced very large pebble, cobble, and boulder deposits including an approximately 15 square mile area of coarse gravel dunes near Spirit Lake, Idaho. The gravel dunes near Marlin, Washington are approximately 6.5 feet high and 200 feet apart, and probably formed in response to 200 foot deep flood waters. The largest Glacial Lake Missoula discharges likely occurred through the Spokane Valley-Rathdrum Prairie area, resulting in as much as 500 feet of flood deposits over an area of about 350 square miles (O'Connor and Baker 1992). Peak discharge through the valley is estimated at approximately 600 million cubic feet per second (O'Connor and Baker 1992).

Glacial lake outburst flooding on a variety of scales occurred in other areas of the United States that were at the ice margin during the Wisconsin Glaciation. For example, a gravel aquifer is associated with glacial flooding along the Umatilla River in Milton-Freewater, Oregon. Some of these regions are further discussed in Chapter 3.

2.5 Hydrogeologic Barriers

A hydrogeologic barrier is defined as the physical, biological, and chemical factors, singularly or in combination, that prevent the movement of viable pathogens from a contaminant source to a water supply well. Where present, a hydrogeologic barrier may protect wells screened in sensitive aquifers (i.e., those in karst, fractured bedrock, and gravel aquifers) from microbial contamination. States have the option to investigate and verify the presence of an adequate hydrogeologic barrier for sources located in hydrogeologically sensitive settings. If a thorough investigation proves that the proposed barrier is protective of the well, then the State may use that information in making further decisions such as evaluating the need for assessment source water monitoring.

A confining unit, a common example of a hydrogeologic barrier, is a low permeability subsurface stratigraphic layer that overlies an aquifer and acts to prevent significant infiltration to the aquifer. Low permeability strata often consist of unconsolidated clay or silt, or their consolidated counterparts, shale and siltstone, but they may also consist of other lithologies (USEPA 1991b). For example, certain glaciated areas of the United States are underlain by cemented till, which is relatively impervious and acts as a hydrogeologic barrier. In order to prove that a given well is protected by a confining unit or other hydrogeologic barrier, it will generally be necessary to conduct a thorough investigation of the surrounding area. A successful investigation will usually need to include evidence that the hydrogeologic barrier for a specific well protects at a minimum the aquifer over an area that includes the zone of influence when the well is pumping.

Confining layers may be discontinuous. They may also be breached by natural processes (e.g., fractures) or anthropogenic activities (e.g., improperly constructed or abandoned wells). Confining layers are rarely absolutely impermeable. Instead, a confining layer is simply characterized by a hydraulic conductivity that is orders of magnitude lower than that of an adjacent aquifer. Confining layers may, in fact, be leaky in that they slowly transmit water from one aquifer to another.

Other environmental conditions that may act to prevent the movement of viable pathogens to an aquifer include the following:

1. Sufficiently long subsurface horizontal or vertical ground water travel times (especially due to vertical flow through a thick unsaturated zone) so that pathogens become inactivated as they travel from a source to a public water supply well.

2. Site-specific physical and chemical (and perhaps even biological) properties of the aquifer and ground water which may serve to decrease the longevity of particular microbes, increase their adsorption to aquifer material, or otherwise decrease the rate at which they are transported to a public water supply well.

It is important to emphasize, however, that the geochemical factors which affect virus and bacteria fate and transport are complex, poorly understood, and vary significantly with virus and bacteria type. Furthermore, in sensitive hydrogeologic settings, pathogens have fewer opportunities to interact with aquifer material as compared with other hydrogeologic environments, such as those consisting of fine-grained, unconsolidated sediments. The potential for biological predation to provide a hydrogeological barrier to pathogen contamination is discussed further in section 3.5; however, there is little documentation of such predation on waterborne pathogens in the scientific literature. Thus, the use of geochemical or biological conditions as a basis for determining an adequate hydrogeologic barrier is present is likely to require detailed, site-specific field investigations. Continuous, non-leaky confining units or very thick unsaturated zones are more likely to be proven adequate hydrogeologic barriers.

The presence or absence of a confining layer is sometimes difficult to determine. The office procedures for determining presence or absence of a confining layer protecting a sensitive aquifer apply equally to determining the presence or absence of a confining layer protecting a shallow aquifer of any kind. The following text provides information to differentiate an unconfined from a confined aquifer.

There is no established permeability (or hydraulic conductivity) range for confining strata. Low permeability rocks typically have permeability values below 10^{-3} cm/sec (10^{-5} ft/sec). Permeability of a confining unit is typically three orders of magnitude lower than the permeability of the producing aquifer (Freeze and Cherry, 1979). Confining layers can be extremely variable in composition so that permeability (and confining performance) can vary significantly in all directions. In general, confining layers that are formed by deposition in open marine environments are the most homogeneous type of confining bed (USEPA 1991b).

Rates of vertical leakage are an important consideration in differentiating highly confined from semi-confined aquifers (USEPA 1991b). Rates of vertical leakage can be calculated using Darcy's Law. Information needed to perform the calculation include: water level for the confined aquifer; water level at the water table; vertical hydraulic conductivity; and confining bed thickness. The equation is presented in EPA (1991b) as equation (2). Unlike horizontal hydraulic conductivity data for most Darcy's Law applications, vertical hydraulic conductivity data are often very difficult or expensive to obtain.

Vertical travel time calculations may be used to differentiate semi-confined from highly confined aquifers (USEPA 1991b). However, the method is not appropriate for identifying confining beds and is difficult to implement. Implementation problems arise because little data is generally available on porosity and vertical permeability of confining beds. Other data, less difficult to obtain, is also necessary, such as confining bed thickness and hydraulic gradient across the confining strata. Because these data are not typically available, the travel time method (EPA, 1991b)(see Appendix B) should be conducted together with other verification methods, such as age-dating with tritium analyses (see Appendix A), to reduce overall uncertainty.

EPA (1994) identifies 14 indicators of confinement and the characteristics used to identify the presence of a confining layer (measured in the confining layer or in the aquifer, as specified). More detailed discussion about each of the 14 indicators is presented in EPA (1991b). Some of the methods described in EPA (1991b) may not be appropriate for confined karst or fractured bedrock aquifers. More detailed information is necessary to apply these methods to a particular hydrogeologic setting. The fourteen indicators are:

1. Geologic maps and cross-sections showing the presence of a continuous, suitable confining layer such as clay above the aquifer.

2. Water level elevation above the top of the aquifer, as measured in a single well screened in the aquifer.

3. Hydraulic head differences between two (with one being karst) aquifers as measured in wells cased to and open in the differing aquifers.

4. Water level fluctuations in the aquifer as the result of barometric or tidal effects but with no response to infiltrating precipitation and recharge.

5. No changes in water level in the aquifer in response to large pumping stress and diurnal water level fluctuations.

6. Pump test with storativity value for the aquifer calculated to be less than 0.001.

Note: Storativity values for confined aquifers may range from 0.005 to 0.00005; much lower than values for unconfined aquifers (which range from 0.01 to 0.30) (Freeze and Cherry, 1979). The concept of storativity was originally developed for analyzing well hydraulics in confined aquifers. It is defined as the volume of water that an aquifer releases from storage per unit surface area per unit decline in hydraulic head due to pumping. Karst limestone or fractured bedrock aquifers do not generally release much water from storage, even when unconfined. Because these aquifers may have very low storativity values even when unconfined, storativity may not be useful for the purposes of identifying confining layers in these aquifers.

7. Leakage pump test (in the aquifer) plotted as drawdown versus time matches analytical solutions (calculated leakage less than 10-3 gal/day/ft2). However, calculation of leakage values for well fields is not routine and there may be limited information available.

8. Three-dimensional numerical ground water flow model with good parameter input data (and appropriate number of nodes) showing low leakage across confining layer.

9. Water chemistry in the ground water indicative of long distance from recharge.

10. No anthropogenic atmospheric tracers such as detectable tritium or fluorocarbons in the ground water. EPA (1991b) notes that tritium analyses may be inappropriate for the case of a confined limestone aquifer, where horizontal flow may be fast enough that ground water contains tritium from lateral recharge and not from vertical leakage through the confining bed.

11. Isotope chemistry showing carbon-14 dates greater than 500 years in the ground water.

12. No detectable contaminants in the ground water identified based on inventory of contaminant sources.

13. Long-term head decline due to pumping does not cause accompanying water chemistry changes in the ground water indicative of vertical leakage.

14. Time of travel through confining strata that exceeds 40 years, where the travel time calculations are based upon hydraulic head gradient, porosity, and hydraulic conductivity measurements or estimates for the confining layer.

EPA (1991b) provides recommendations for the methods that are most appropriate for evaluating confinement. The most important recommendation is that the determination be based on an integration of geologic, hydrologic, and hydrochemical approaches. The geologic approach is necessary to determine whether there is a confining bed and whether there are pathways through that bed. The hydrologic and hydrochemical approaches document whether there is actually leakage through the confining bed. Collecting both hydrologic and hydrochemical data allows for a comparison of the results from one approach with the results from another. Of the available hydrologic methods, those based upon water level data (including continuous recorder data) and potentiometric surface data are most useful as such data are the easiest and least expensive to obtain (USEPA 1991b). Of the hydrochemical data, tritium analyses are the most useful (USEPA 1991b).

The Groundwater Section of the Illinois Environmental Protection Agency has developed a list of five diagnostic properties to determine if a well is protected by a confined aquifer

(ILEPA 1995). According to the list, wells are most likely to be protected if they satisfy the following criteria:

1. At least one contiguous unit of impermeable geologic materials greater than 10 feet thick overlies the aquifer (excluding the top 10 feet of soil materials).

2. The top of the uppermost aquifer is greater than 50 feet from the surface.

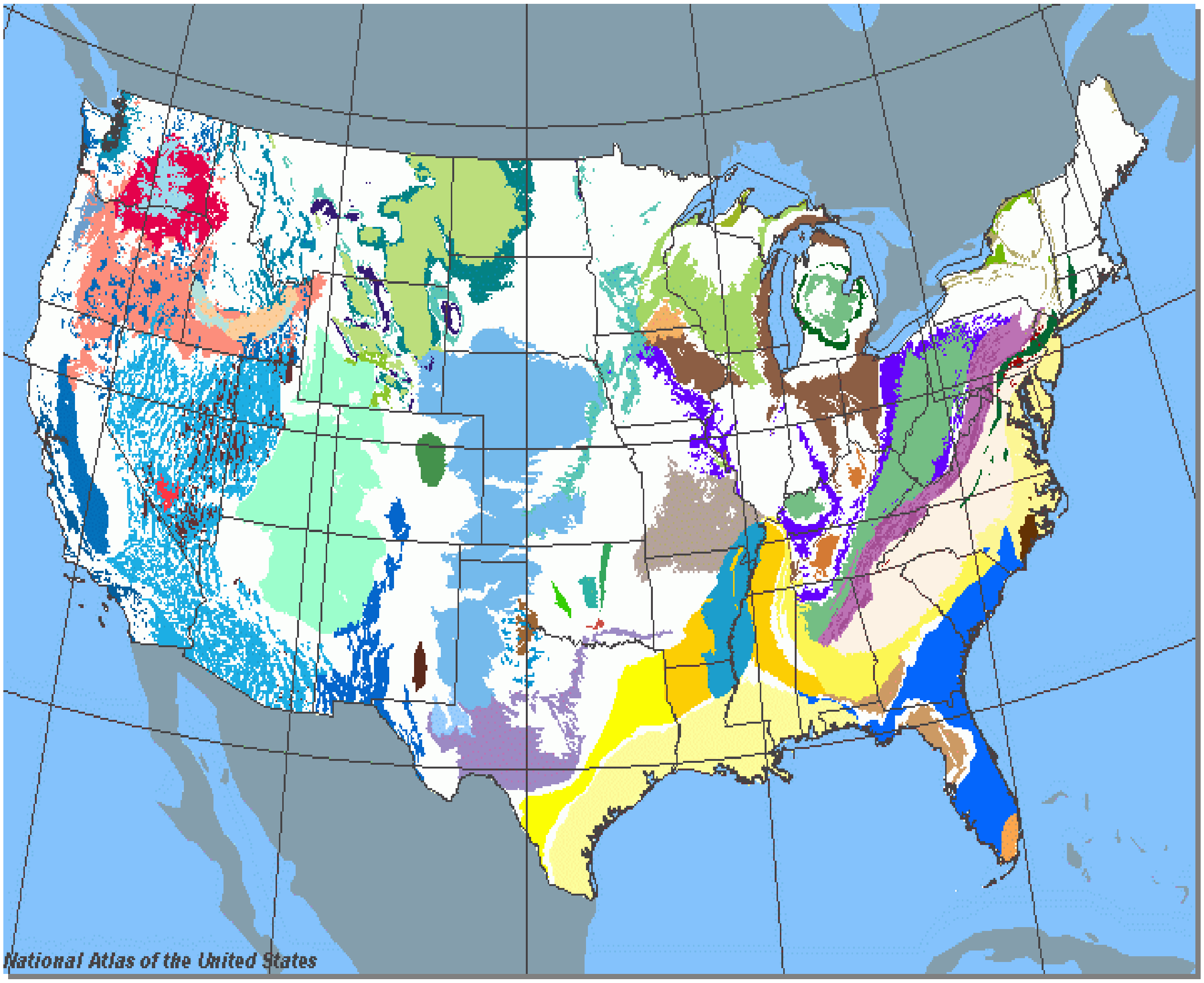
3. The static and pumping water levels of the PWS are above the top of the aquifer (using the most recent data).

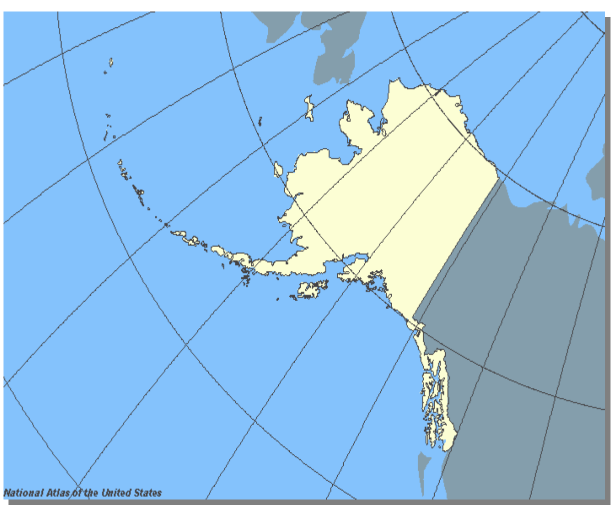
4. The well is located in an upland (i.e., non-alluvial/outwash) geologic setting.

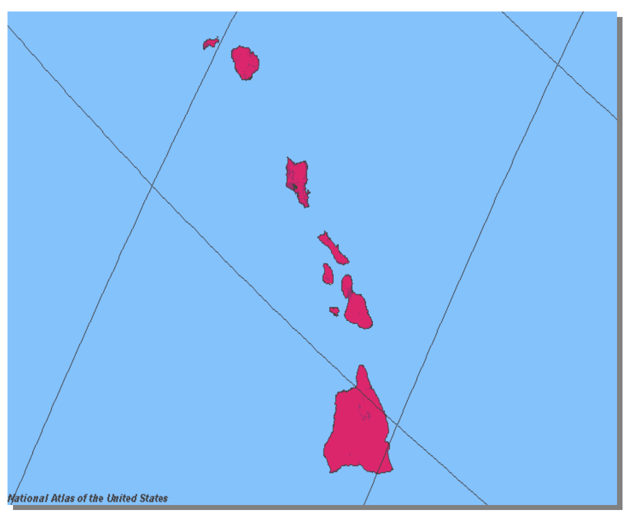
5. The storativity value for the aquifer is less than or equal to 0.001.

The Illinois Environmental Protection Agency uses the above criteria in a weighted scoring process in determining whether or not an aquifer is confined. Out of a total possible score of 10 points, criteria 1 and 2 are weighted with 3 points each; criterion 3 is worth 1 point if just the static water level is above the top of the aquifer and 2 points if both the static and pumping water levels are above the top of the aquifer; criteria 4 and 5 are worth 1 point each. If a public water supply well meets criteria 1 and 2, and also amasses at least 6 points in the scoring process, it is determined to be protected by a confining unit. Exhibit 2.4 provides a national perspective of aquifers and their locations.

Exhibit 2.4 Aquifers of the United States







Source: National Atlas of the United States, March 5, 2003, http://nationalatlas.gov

3. Hydrogeologic Sensitivity Assessments

As discussed in section 1.1, States may use hydrogeologic sensitivity assessments (HSAs) as a tool to determine if GWSs are obtaining water from hydrgeologically sensitive settings. Desktop HSA approaches are emphasized in this guidance document because, in most cases, States can determine a PWS's aquifer type without conducting a field investigation. Field investigations can be conducted if desktop analyses provide insufficient information to complete an HSA. Because a desktop analysis depends on the availability of reliable data and information, hydrogeologic data sources are discussed first. The desktop HSA approaches are then presented, and case studies are included to demonstrate how data generated through Source Water Assessment Program (SWAP) efforts and/or other water resources investigations can be used to support HSAs. In particular, section 3.3.2 demonstrates how in the City of Twin Falls, Idaho, a Source Water Assessment Report for the Blue Lakes Well Field, a community drinking water system, provides *all* of the necessary information for completing an HSA. Similarly, the Source Water Assessment Program provides documents regarding the community water system in Trenton, Kentucky (discussed in section 3.2.2), which also contain all of the information necessary to complete an HSA.

This document includes three aquifer types as sensitive: karst, fractured bedrock, and gravel. Such designations allow States to simply identify the aquifer type of a ground water source for sensitivity assessments, instead of conducting detailed hydrogeologic investigations to characterize each source's ability to rapidly transmit potential contaminants. A State can designate other sensitive hydrogeologic settings if it chooses. States can also choose to demonstrate that a hydrogeologic barrier is protective of a public water system (PWS) well by showing that the sensitive aquifer is adequately protected by the barrier over an appropriate area surrounding the well.

Among the many risk factors applicable to assessing the likelihood for fecal contamination of a PWS well, one risk factor, sensitive aquifers, is perhaps most important. To illustrate this importance, consider a geologic map of the Michigan basin. Such a map can be created at http://nationalatlas.gov. Click "Map Maker." Click "Geology." Click "Geologic Map." Then click on the area of the map to be zoomed (Michigan, in this case). The map shows a large geographic area extending from southern Ohio to northern Michigan and northern Ontario. The boundary between rocks of Silurian and Devonian age is marked as an arc that extends from Ohio to Ontario to Michigan. As one traces the boundary with a finger, it passes over the locations of three PWS waterborne disease outbreaks. Despite being widely separated, these three outbreaks have one common feature; they all are located in wells that produce from the Upper Silurian Bass Island Formation, a karst limestone aquifer. The outbreaks in South Bass Island, Ohio, Walkerton, Ontario and Drummond Island, Michigan are not random occurrences. Rather, they are associated with a specific aquifer type. Another example is found in Texas, where outbreaks in New Braun, Georgetown, and Brushy Creek, Texas are all associated with the karst limestone Edwards Plateau aquifer. Because karst limestone and other sensitive aquifers represent a significant risk factor, this document provides additional information to identify sensitive aquifers.

3.1 Identifying Aquifer Types

The geologic and hydrogeologic characteristics of karst, fractured bedrock, and gravel aquifers are reasonably well known. Such knowledge allows the identification of sensitive settings using high resolution data and a desktop analysis. This section outlines some diagnostic characteristics and general approaches for identifying each sensitive aquifer type without conducting field investigations. If a desktop analysis is not possible due to limited data availability, the diagnostic characteristics noted below for each aquifer type can help guide the types of field investigations that are needed to make a sensitivity determination. Field investigations are likely to be needed only on rare occasions. Karst aquifers comprise a large percentage of all productive aquifers in the United States. When considering hydrogeologically sensitive aquifers only, karst aquifers make up an even larger percentage of the category. Thus, this guidance manual provides more detail on the complex flow regimes and diagnostic characteristics of karst regions than it does for the other two sensitive aquifer types (fractured bedrock and coarse gravel deposits). Karst aquifers are the sensitive aquifer type most likely to be encountered by the largest number of water systems. Nevertheless, karst, fractured bedrock, and gravel aquifers are considered equally sensitive. Each of these aquifer types may pose a risk of pathogen contamination to PWS wells located within them.

3.2 Karst Regions and Aquifers of the United States

This section discusses U.S. karst regions and aquifers in general, although only those developed in limestone bedrock are considered sensitive. On the other hand, potentially soluble rocks such as dolomite and evaporites are not explicitly considered sensitive. Again, States may choose to consider all carbonate and other soluble rock aquifers as possibly karstic and designate them sensitive.

A map of the United States showing karst regions was presented by Davies et al. (1984) at a scale of 1:7,500,000. Detailed descriptions of the major U.S. karst regions accompany the map. A list of significant karst aquifers was also published by EPA (1997).

About 20 percent of the United States is underlain by karst aquifers of various types. Karst aquifers underlie almost 40 percent of the United States east of the Mississippi River (Quinlan 1989). Where soluble rocks such as limestone and dolomite are present at or near the surface, solution openings are common. A region's topographic relief, soluble rock thickness, and hydrogeology determine the depth to which solution openings occur. For example, the vertical extent of solution openings is known to be as great as 1,100 feet in mountainous areas of the western United States. In the eastern United States, where topographic relief is lesser, depths are generally less than 400 feet, with a maximum of 650 feet. Beneath many broad river valleys throughout the country, solution features in carbonate rocks extend to depths of approximately 100 feet (Davies et al. 1984).

In the region of the United States that was formerly covered by Pleistocene ice sheets (glaciers), karst-related caves and fissure openings may have been partially filled by glacial debris. The southernmost advance of the ice sheets covered New England, New York, northern New Jersey, northeastern and northwestern Pennsylvania, most of the States bordering the Great Lakes, and much of the land north of the Missouri River (Davies et al. 1984). Because of glacial erosion or filling, caves in glaciated terrain typically have lengths of less than 1,000 feet. South of the formerly glaciated areas, caves and other solution features are more identifiable, and in general, the number and size of solution features increases with decreasing latitude. Furthermore, the number of solution openings varies according to the age and structure of the soluble rocks. More deformed or folded rocks (deformed as part of mountain building processes) may have fewer solution openings than younger, undeformed soluble rocks. Solution openings are most highly developed in Mississippian or younger limestones (Davies et al. 1984).

The carbonate bedrock may be present at (or directly beneath the soil cover of) the land surface in many karst regions (Winter et al. 1998). The Edwards Aquifer in south-central Texas is an example of such a region. Across this area, enlarged fractures, solution cavities along fractures, and sinkholes are present at (or very near) the land surface and extend down into the bedrock. Precipitation in such karst regions (particularly when the precipitation falls on outcropping bedrock) tends to infiltrate the land surface rapidly, seeping downward and acting as a source of recharge to the underlying aquifer. A considerable amount of recharge to karst aquifers can also be provided by streams that flow across local karst features. Water in karst region streams can flow out of the base of the stream as it encounters the karst features, rapidly infiltrate downward, and thereby be lost from the surface as it recharges the underlying aquifer (i.e., a “losing stream”). Even the largest streams that originate outside a karst region outcrop area can be dry within the outcrop belt for most of the year (Winter *et* al. 1998).

In other karst regions, the karstic limestones or dolomites are covered with a veneer of material that can obscure some of the karst features. This is referred to as a “mantled karst.” An example is northwest Arkansas, a region characterized by many springs, seeps, losing streams, and a regolith mantle of variable thickness (Brahana J.V. et al. 2002). As karst solution features enlarge in subsurface limestones or dolomites, the underlying bedrock support is removed, and the overlying cover or mantle material can slump into the karst solution features. In these cases, even if the mantle material may act as a confining unit, the resulting slumpage breaches the confining unit and water can readily infiltrate the underlying karst aquifer.

Mantled karsts are quite susceptible to slumpage. In Florida, most land-surface depressions containing lakes formed by the slumpage of unconsolidated surficial deposits into sinkholes that are caused by dissolution of the underlying limestone (Winter et al. 1998). Shallow ground water can quickly flow through such a lake and into the underlying karst aquifer. In this way, ground water flowing through mantled karst aquifers may transport fecal contamination from surficial water bodies to wells. In northwest Arkansas, samples taken from springs flowing from mantled karst were found to be contaminated with fecal coliform (Whitsett et al. 2001).

3.2.1 Diagnostic Characteristics

Although lithology cannot be used to definitively identify whether or not an aquifer is karstic, the presence of carbonate rocks, especially when subareally exposed in a humid environment, generally indicates the existence of a karst aquifer (USEPA 1997). For the purpose of simplifying the HSA process, this document presumes that all limestone aquifers are karstic, and therefore sensitive, while presuming that all other soluble aquifers are not karstic and not sensitive. The information in this section may be useful to States because they may *choose* to designate karst aquifers developed in other carbonate or soluble rocks (e.g., dolomite and gypsum) as sensitive. Furthermore, the characteristics diagnostic of karst regions and aquifers may be useful for HSA determinations in cases when a State may not have definitive information on a given PWS's lithology (i.e., whether it is limestone).

This section presents geomorphic, geologic, and hydrogeologic characteristics of karst regions and/or aquifers. This information may be useful for conducting desktop hydrogeologic sensitivity assessments. Information that can be gleaned from maps, aerial photos, and other spatial data sources are discussed first, followed by a brief discussion of how karst hydrology may manifest itself in a variety of existing hydrogeologic data sources and reports. Distinctive karst landforms produced by the dissolution of underlying carbonate bedrock may be the first clue that a locality's PWS is drawing water from a karst aquifer. Nevertheless, site-specific hydrogeologic data is the most direct evidence.

EPA (USEPA 1997) suggests that karst regions can be identified by the presence of one or more of the following land surface features: sinkholes; springs; caves; sinking or losing streams; discontinuous drainage networks; and dry valleys in humid climates. Field surveys are the most reliable way to identify karst features, but many are also recognizable on sufficiently detailed aerial photos or topographic maps. Stereoscopes (optical instruments used to create three dimensional images from stereoscopic aerial photographs, see section 1.5.5) can be used to facilitate interpretations of stereoscopic aerial photographs. Identifying karst features using topographic maps may be more challenging. Nevertheless, contour lines representing closed surface depressions (sinkholes) and discontinuous drainage networks (including disappearing and reappearing streams) are indicative of a karst landscape (see Exhibit 2.2). Local names for natural features may also be noted on a map and can provide clues that an area is influenced by karst hydrology. For example, the name "Lost River" is often a good indication that the stream in question is located in a karst landscape, and disappears into the subsurface.

Geologic and geomorphic characteristics diagnostic of karst regions and/or karst aquifers may also be described in existing data and reports. For example, the following features are indicative of karst terrain and may be noted in reports describing the field reconnaissance of a particular area: dissolution-enlarged fractures or bedding surfaces; karren (dissolutional, subaerial, water-carved grooves in rock, commonly subparallel); and grikes (soil-filled, dissolutionally-enlarged fractures or grooves; also known as cutters or soil karren) (USEPA 1997). These features may be seen in outcrops or road cuts, or encountered through drilling.

Hydrogeologic data compiled and described in existing reports may also reveal characteristics indicative of karst hydrology. For example, a karst aquifer may be identified using aquifer pump test results for a well penetrating, and open in, the aquifer. Continuous pumping data showing stepped drawdown in the pumped well, or one or more nearby observation wells, suggests that there is flow through conduits or fractures. However, the converse (a smooth drawdown curve) does not always indicate a non-karstic aquifer because karst aquifers are highly variable (USEPA 1997). Stepped drawdown may not be observed in a well that does not intersect sufficient numbers of solution-enlarged fractures or conduits, although drawdown data collected for another well nearby may show entirely different results.

EPA (1997) suggests that the following hydrogeologic characteristics, possibly documented in existing data and/or reports, can be used to identify karst (or fractured) aquifers:

- Stepped drawdown during continuous pumping.
- Irregular cone of depression, or no cone of depression, as defined by multiple observation wells around a pumped well.
- Drawdown plotted against discharge indicates non-linearity.
- Stair-stepped, irregular configuration of potentiometric surface.
- Bimodal hydraulic conductivity data (when plotted as the logarithm of hydraulic conductivity) for a suite of wells completed in the same formation. Note that unimodal hydraulic conductivity is also possible in a karst aquifer.
- Differing water levels in closely adjacent wells.
- Bimodal or polymodal distribution of temporal specific conductance data from a given well.
- Significant spatial variation in specific conductance and hydraulic conductivity as interpreted from well logging devices.
- Significant variation in well discharge distribution during constant rate pumping.
- Significant variations in the distribution of flow with depth, for a pumped or unpumped well, measured using electromagnetic or thermal flow meters.
- Significant differences in tracer breakthrough depending on the location of the injection well (given recovery at the same well).

3.2.2 Desktop Approaches

It is feasible to determine a PWS well's aquifer type without field investigation in most cases. A variety of hydrogeologic data sources (discussed in section 1.5) may provide the necessary information. In some cases, one data source may suffice (e.g., an existing study of the PWS well of interest may identify the well's aquifer type). Nevertheless, determining aquifer type in a desktop analysis will often require multiple data sources used together. This guidance document recommends a simple step-like approach to determining aquifer type where the most directly relevant information is used first (e.g., primary data sources such as driller's logs, aquifer maps, geologic maps, and existing hydrogeologic investigation reports). If such data sources are insufficient or unavailable, an HSA can progress to using less direct information (e.g., soil surveys, topographic maps, and aerial photos) to facilitate an aquifer type determination.

This section illustrates the step-like HSA approach using two case studies. The first case study shows how primary information gathered in a wellhead protection study (i.e., a driller's log) identifies the aquifer type for an undisinfected well. Similarly, the second case study illustrates how information available from State SWAP reports can be used for an HSA. EPA is encouraging States to use information from their source water assessment efforts (i.e., through their WHPPs and SWAPs) as they conduct their HSAs, and to coordinate efforts among these programs wherever possible.

Case Study #1 – Fincastle, Virginia

Fincastle, Virginia is located in Botetourt County, a mostly rural area of approximately 30,500 residents near Roanoke, Virginia. The Town of Fincastle, Botetourt County's historic county seat, owns and operates a small PWS that served 975 people using two wells at the time of the study described below. Rapid growth in this region is likely to have placed additional demands on the PWS. The PWS wells were constructed in the mid-1970's after rural private wells that pumped from the region's shallow aquifer experienced hydrocarbon and fecal contamination. The two PWS wells tap a deeper aquifer that was not affected by the shallow aquifer's contamination. Available data for these wells include design and construction information (e.g., well depth and diameter), a driller's log, pump tests, and well yields. The PWS's service area includes residences, businesses, government offices, churches, a nursing home, and three schools (Virginia DEQ 1993).

A wellhead protection study for the two Fincastle wells identified farm animal wastes and residential septic systems as potential fecal contaminant sources. The study also noted that State regulations allow septic systems to be located as close as 75 feet from a PWS wellhead (Virginia DEQ, 1993). One of the Fincastle PWS wells is located 100 yards from a commercial gas station and across the street from a farm that obtained a permit for a septic system in the early 1990s (Virginia DEQ 1993). Clearly, potential sources exist for fecal contamination of the Fincastle PWS wells.

The first well, Fincastle #1, is drilled 400 feet into dolomite and limestone. Fincastle #2 penetrates to a depth of 475 feet in the same rock types. Both wells intersect solution cavities and water-bearing fractures according to the wellhead protection study (Virginia DEQ, 1993):

> The Driller's Log indicates two caves or openings with muddy water were encountered from a depth of 125 feet to 130 feet and from 190 feet to 195 feet, and both of these openings were cased off. The material from a depth of 195 feet to 470 feet was described simply as limestone with "broken limestone" zones from 405 feet to 407 feet and from 428 to 431 feet. A cave opening was reported at the bottom of the well, for the depth interval 470-475 feet (Virginia DEQ 1993).

Based on the driller's log, the aquifer is determined to be karst. Additional information from other sources, such as the Virginia geologic map and the Botetourt County Soil Survey, help confirm the determination of this well's hydrogeologic setting.

According to the Virginia Department of Health Environmental Engineering Field Office in Lexington, Virginia, Fincastle well #1 is not chlorinated. Fincastle well #2 is chlorinated. An undisinfected well located in a sensitive hydrogeologic setting, such as the limestone and dolomite karst from which both Fincastle wells are pumping, should be considered for source water assessment monitoring. If the chlorination system at Fincastle well #2 does provide a four log virus inactivation, it should also be considered for fecal indicator analysis.

Case Study #2 – Trenton, Kentucky

The City of Trenton, Kentucky operates a community water system (CWS) that serves 868 people (KYDEP 1999). The CWS has an average daily withdrawal of 110,000 gallons and is supplied by three ground water production wells. Two of the wells were constructed in 1900 and 1936, respectively, and were probably the wells for Trenton's first public water supply. The third well was completed in 1992 (KYDEP 1999).

The State of Kentucky has completed a Phase I WHPP for the City of Trenton (KYDEP 1999), as part of its Source Water Assessment Program. The WHPP notes that the CWS's three wells are drilled in a karst aquifer, and more specifically, a hydraulic conduit (solution opening), and includes a general discussion of the regional geology and hydrology. The report provides farther hydrogeologic details, most notably a well log for well #3 (drilled in 1992), indicating that the city's wells are drawing from a limestone karst aquifer. The well #3 driller's log indicates that limestone bedrock is encountered at 51 feet. At 84.5 feet, a void is recorded that continues to a depth of 85.5 feet. Limestone is encountered below the void to a depth of 90 feet, which is the well depth, and the well log records "good water" for this interval (KYDEP 1999).

Drillers logs are not available in the Phase I WHPP for the City of Trenton's two other wells. However, Attachment 3 of the report includes water well inspection records for wells #1 and #2. These records indicate that all three of the City's wells are in very close proximity and that wells #1 and #2 are both drilled to a depth of 85 feet. Comparing these depths to the well #3 driller's log indicates that these two wells are indeed drawing from a void (solution opening) in limestone karst, supporting the narrative information found in Attachment 2.

Although this CWS is discussed here as a hypothetical case for preparing an HSA, the desktop analysis would (hypothetically) designate the aquifer from which this system's three wells pump as a sensitive aquifer.

3.3 Fractured Bedrock Regions and Aquifers

All igneous and metamorphic aquifers are considered fractured bedrock aquifers, and therefore sensitive aquifers for the purposes of an HSA. Unlike karst hydrogeologic settings, regions underlain by fractured bedrock generally do not have characteristic topographic expressions or landforms. An exception may be domal highlands with thin soils surrounded by flat topography, examples of which include the Adirondacks in New York, the Llano uplift in Texas, South Dakota's Black Hills, and the St. Francis Mountains of Missouri. Fractured bedrock regions may have mountainous (e.g., the Sierra Nevada) or gently rolling topography such as the metamorphic Piedmont region of the eastern United States. Therefore, topography is not necessarily a diagnostic tool for identifying fractured bedrock regions. Instead, States are encouraged to use geologic maps to identify their igneous and metamorphic fractured bedrock regions. These rocks may be exposed at the surface or, in the previously glaciated regions of the

United States, covered by a relatively thin veneer of sediment. In a few regions of the United States, fractured bedrock is known to be overlain by sedimentary rocks, but is shallow enough to be an aquifer. Sedimentary strata of Cambrian or Ordovician age, identifiable on geologic maps, may thinly cover igneous or metamorphic bedrock in a few places in the mid-continent.

3.3.1 Diagnostic Characteristics

This document presumes that any igneous or metamorphic aquifer is a fractured bedrock aquifer and therefore sensitive. Therefore, the characterization of an aquifer that feeds a given PWS well as igneous or metamorphic is sufficient to complete an HSA. Available data and reports, however, may reveal (in cases where lithology is uncertain) additional aquifer characteristics that are diagnostic of fractured bedrock aquifers and could support an HSA aquifer type determination.

EPA (1991a) suggests that the following data may be useful as part of the characterization of an aquifer as a fractured versus a porous medium. Although this data can be collected in the field when necessary, in many cases it will already be available to States through the desktop data sources discussed in sections 1.5 and 3.2.2.

1. Discharge-drawdown plots using aquifer pump test data.

Hickey (1984, cited in USEPA 1991a) suggests using an aquifer pump test with incrementally greater discharge rates in the pumping well, accompanied by drawdown measurements in observation wells measured at one-hour increments, to test whether an aquifer is fractured or porous media. Plotting discharge versus drawdown on an arithmetic scale, a non-linear fit to the data may suggest that fracture flow is occurring.

2. Time-drawdown plots using aquifer pump test data.

Time-drawdown curves for observation wells located in two or more different directions from the pumped well that have different shapes or sharp inflections may be indicative of fractured bedrock aquifers.

3. Contour map showing points of equal drawdown.

Maps showing lines of equal drawdown (drawdown contours) are compiled using drawdown values measured at multiple observation wells. Linear or irregular drawdown contours, rather than circular or elliptical contours, are indicative of fractured bedrock aquifers. Furthermore, Risser and Barton (1995) suggest that if water levels in multiple observation wells decline, but the response is greatest at some distant well, then the aquifer may be a fractured bedrock aquifer. Also, if water levels in some observation wells do not decline in response to pumping, while levels in other nearby wells decline, a poorly connected fracture network may be present.

4. Water table surface configuration.

A “stair-step” water table configuration could be indicative of a fractured bedrock aquifer. Such configurations occur in sparsely fractured rocks when there are large contrasts in hydrogeologic properties between massive blocks and the fracture zones that bound them.

5. Hydraulic conductivity distribution.

Aquifers with strongly bimodal hydraulic conductivity distributions may be fractured bedrock aquifers.

Black (1989), as cited in Risser and Barton (1995), cautions that tests to determine whether an aquifer is fractured can provide misleading results because fractured bedrock aquifer characteristics can change with time. For example, a non-linear response may occur at the beginning of a well pump test, indicating flow from a single, planar, vertical fracture (i.e., one-dimensional flow). As the pump test continues, horizontal fractures begin to contribute to the flow. The flow pattern becomes more two-dimensional, and a radial response in the aquifer may result. Thus, a pump test analysis relies on evaluation of the early pumping results to determine if the aquifer is fractured.

3.3.2 Desktop Approaches

As noted above, this guidance document recommends a simple step-like approach to determining aquifer type using the most directly relevant information first (e.g., primary data sources such as driller’s logs, aquifer maps, geologic maps, and existing hydrogeologic investigation reports). If primary data sources are unavailable or do not definitively identify whether an aquifer is composed of igneous or metamorphic rocks, less direct information may help to confirm an identification and complete an HSA. This section illustrates the simple step-like approach for completing an HSA for PWSs located in a fractured bedrock setting. Case Study #3, below, demonstrates how State source water assessment program data can be integrated with other existing data, in this case a State-wide bedrock geologic map, to determine aquifer type. Case Study #4, on the other hand, demonstrates that in some cases, a source water assessment report *alone* can be used to complete an HSA.

Case Study #3 – Enfield, New Hampshire

The State of New Hampshire is underlain almost entirely by igneous and metamorphic bedrock. In some places, the bedrock is at or very near the surface. However, glacial overburden of variable thickness is widely distributed throughout the State. Ground water PWSs in New Hampshire often draw water from glacial overburden aquifers, but some draw from igneous or metamorphic bedrock aquifers (i.e., fractured bedrock aquifers).

Enfield is a small, rural town in western New Hampshire located approximately 10 miles east of the Connecticut River, which forms the border with Vermont. The Enfield Water Department operates four active wells serving 1,125 people. The State of New Hampshire has completed a source assessment report (SAR) for the Enfield Water Department (NHDES 2001a). The SAR indicates that all four of the PWS’s wells are bedrock wells, and that three of them have well depths of at least 425 feet. Well depth is not reported for one of the wells. A GIS map accompanying the SAR locates three of the wells in the northern part of Enfield, and one well is shown farther northwest and just across the town line in nearby Canaan (NHDES 2001a). Consulting the State's 1:250,000 scale bedrock geology map indicates that the area in the vicinity of the wells (and the whole region) is underlain by igneous and metamorphic rocks (Bothner and Boudette 1997). Therefore, given the fact that the wells are bedrock wells, and the GWR's presumption that igneous and metamorphic aquifers are fractured bedrock aquifers, Enfield's public water supply wells are drawing water from a fractured bedrock (and thus sensitive) aquifer.

Case Study #4 – Blue Lakes Well Field, City of Twin Falls, Idaho

The City of Twin Falls, Idaho has a community drinking water system consisting of 10 ground water source wells, four of which comprise the Blue Lakes Well Field, north of the Snake River in Jerome County. These four wells are joined together in a manifold. Although total coliform bacteria were detected in the water distribution system on four occasions between 1994 and 1998, no microbial contaminants have ever been detected in the samples that were collected from the well manifold according to the Idaho Department of Environmental Quality (IDEQ) (IDEQ 2002a).

According to IDEQ (IDEQ 2002a), the Blue Lakes wells draw from a regional aquifer consisting of highly fractured, layered basalts of the Snake River Group. These basalts host one of the most productive aquifers in the United States, often yielding up to 3,000 gal/min for wells screened in only 100 ft of the sometimes 5000 ft. thick flows. The aquifer is unconfined over most of its areal extent, although interbedded clays and small areas of less fractured basalt may create confined conditions in some localized areas. The aquifer is recharged by surface water irrigation, stream losses, direct precipitation, and underflow from tributary basins. According to well logs, the aquifer is 500 to 1500 feet thick in the region of the Blue Lakes wells, and is overlain by 1 to 23 feet of sediment. In the area immediately surrounding the wells, the water table is only about six feet deep. Local area well logs further indicate that the vadose zone is predominantly fractured basalt. The Source Water Assessment Final Report, prepared by the Idaho Department of Environmental Quality, contains all of this information and also indicates that the Blue Lakes wells are only about 20 feet in depth (IDEQ 2002a).

Given that the aquifer from which the four Blue Lakes wells draw water is composed of basalt (i.e., igneous rock), it is considered a fractured bedrock aquifer for the purposes of an HSA. Thus, this aquifer is considered sensitive, assuming the City has not shown that a hydrogeologic barrier exists for these wells.

3.4 Gravel Aquifer Hydrogeologic Settings

Glacial Lake Missoula is thought to have produced some of the largest glacial outburst flood deposits known in North America (see section 2.4). Glacial lakes on a smaller scale (although still large) also formed in other ice margin environments of North America during the Wisconsin period. The southernmost advance of ice sheets during the Wisconsin glaciation covered New England, New York, northeastern and northwestern Pennsylvania, and much of the areas north of the Ohio and Missouri Rivers. Recent research describes coarse gravel deposits in

Wisconsin that may be the result of subglacial-lake outburst flooding from the Laurentide Ice Sheet (Cutler et al. 2000).

Large lakes also formed in some of the interior basins of the Intermountain West during glacial periods. These lakes were not proximal to the margins of the continental glacier, but their formation was related to the climatic conditions that caused, and were perpetuated by, the continental ice sheets. Specifically, increased precipitation, less evaporation, and meltwater from nearby alpine glaciers raised lake levels in many interior basins because they have no outlet to the ocean. An example was Glacial Lake Bonneville, the remnant of which is the modern, and considerably smaller, Great Salt Lake in Utah.

In contrast to Glacial Lake Missoula, flooding from Glacial Lake Bonneville is estimated to have a peak discharge of about 35 million cubic feet per second. Glacial Lake Bonneville floods also produced coarse gravel flood deposits (O'Connor 1993). Bonneville flood deposits may be aquifers along portions of the Snake River in southern Idaho, including areas mapped as the Melon Gravel (Malde and Powers 1972 cited in O'Connor 1993) and the Michaud Gravel in the Pocatello and American Falls areas (Trimble and Carr, 1961a and 1961b, cited in O'Connor, 1993).

3.4.1 Diagnostic Characteristics

As noted in section 2.4, catastrophic floods can produce coarse gravel deposits. These floods are typically associated with the rapid failure of ice-dammed lakes during glacial periods. Coarse gravel deposits can be produced by other processes, such as flash flooding in steep terrain, but these deposits tend to be very small and localized, and are unlikely to form public water supply aquifers. This section, therefore, will focus on some U.S. regions known to have coarse gravel deposits produced by glacial lake outburst flooding or related proglacial outwash processes. States are encouraged to consider the Quaternary depositional history of a region when conducting HSAs for PWS wells screened in unconsolidated aquifers. The Quaternary period is defined by approximately the last 2 million years, during which glacial periods were common. Quaternary glaciers influenced the modern landscape in large areas of North America. If a particular site has a geologic history of glacial lake outburst flooding, it is likely that a coarse gravel aquifer is present, as opposed to an unconsolidated aquifer with significant amounts of fine-grained material.

3.4.2 Desktop Approaches

As noted above, this guidance document recommends a simple step-like approach to determining aquifer type using the most directly relevant information first (e.g., primary data sources such as driller's logs, aquifer maps, geologic maps, and existing hydrogeologic investigation reports). If primary data sources are unavailable or do not definitively identify whether or not an aquifer is composed of coarse gravel sediment, less direct information such as county soil surveys may provide the information necessary to confirm an identification and complete an HSA. This section illustrates the simple step-like HSA approach with a case study demonstrating how State source water assessment program data can be integrated with other existing data to determine aquifer type.

Among unconsolidated (e.g., sand, sand and gravel, gravel) aquifers, only coarse gravel deposits resulting from glacial outburst floods are considered sensitive aquifers for this guidance document. States have discretion to consider any unconsolidated aquifer a sensitive aquifer. As in the above sections, the following case study illustrates how information gleaned from a desktop analysis can be used to determine a PWS well's aquifer type.

In the following case study, most of the relevant information for completing an HSA can be gleaned from the system's source water assessment report in combination with accompanying maps (IDEQ 2000). Determining aquifer type from a desktop analysis may often require the use of additional data sources. Again, EPA is encouraging States to use information from their source water assessment efforts (i.e., through their SWAPs) as they conduct their HSAs, and to coordinate efforts among these programs wherever possible.

Case Study # 5 – Post Falls, Idaho

The surficial geology of Post Falls, Idaho consists primarily of Rathdrum Prairie gravels, deposited by the repeated catastrophic flood releases of Pleistocene glacial Lake Missoula, discussed in section 2.4. Post Falls is located 30 miles downstream from the former Clark Fork ice dam (responsible for forming Glacial Lake Missoula), and 15 miles upstream from the lake where most of the flood waters were channeled (Breckenridge and Othberg, 1998).

The Riverbend Water Company is a non-community, non-transient public water system in western Post Falls, which uses two wells to supply 26 connections with water in a commercial/industrial area (IDEQ 2002b). It is among 186 water systems that draw their water from the Rathdrum Prairie Aquifer. This aquifer is the sole source of water for over 400,000 people (i.e., most of the residents of Spokane County, Washington, and Kootenai County, Idaho).

According to the source water assessment report, the Riverbend Water Company's two wells are screened at around 180 ft and 160 ft from the surface just north of the Spokane River in Post Falls (IDEQ 2002b), which is located in the flood gravel deposits (personal communication, David Risley, Source Water Assessment Program lead, IDEQ). An aquifer atlas that has been sent to all water systems indicates that the entire Spokane-Valley-Rathdrum Prairie Aquifer is composed of thick layers of coarse-grained flood gravel (and cobble and boulder) deposits (IDEQ 2000). The source water assessment report references well logs to show that there is neither a thick unsaturated zone nor a confining unit in the vicinity of the wells (IDEQ 2002b). Thus, the Riverbend Water Company's two public water system wells are good examples of wells screened in a sensitive gravel aquifer, with no hydrogeologic barrier to negate the sensitive aquifer designation.

3.5 Hydrogeologic Barriers

States may wish to consider the presence of a protective hydrogeologic barrier for public water systems (PWSs) in their evaluation of wells drawing water from sensitive aquifers and in any follow-up activities. For example, a limestone aquifer and all wells producing water from that aquifer may be designated as sensitive, but the State may also identify the presence of a hydrogeologic barrier (i.e., confining layer) and may able to demonstrate that the barrier is protective of the aquifer.

All proposed hydrogeologic barriers should be carefully evaluated. For example, it cannot be assumed that all confining layers are protective, are continuous over the area of interest, or have identical properties. Confining layers may be breached by unplugged water or injection wells or may be excessively leaky, allowing rapid transport of fecal contaminants from near-surface environments to the underlying aquifer. It may be possible at some sites to use tracer tests and/or pumping tests, described in Appendix A, to evaluate such situations. The difficulties associated with determining if site-specific geochemical conditions will provide an adequate hydrogeologic barrier were reviewed briefly in section 2.5. In sensitive aquifers with relatively unpredictable ground water flow, the identification of a barrier to pathogen contamination will generally be a technically-based determination that uses as much site-specific hydrogeologic information as possible. The following examples demonstrate why it is important that hydrogeologic barriers be evaluated carefully.

The Floridan aquifer system is the primary drinking water source for the Orlando, Florida area. It is a carbonate aquifer system overlain in the region by a confining unit composed of 150 feet of sandy clay, silt, and shell. The integrity of this protective layer is compromised because Orlando has over 300 stormwater drainage wells that are completed in the upper karstic unit of the aquifer system (i.e., the Upper Floridan aquifer). The operation of these drainage wells is considered necessary for the disposal of excess surface water (including urban stormwater runoff), but the water receives no treatment prior to injection into the aquifer system. Although most of the Orlando area's public supply wells draw water from the deeper Lower Floridan aquifer, separated from the Upper Floridan aquifer by a semiconfining layer, some of the water wells are screened in the Upper Floridan (NRC 1994). Although widespread contamination has not occurred, the Floridan aquifer system in the Orlando area is clearly a sensitive aquifer (because it is limestone), and the 150 ft confining unit is not an adequate hydrogeologic barrier (because it has been breached by the stormwater drainage wells).

In a paper by Johnson et al. (2000), emphasis was placed on the importance of careful investigation of site-specific conditions when evaluating a confining layer. Although a confining unit overlies the Charnock well field in Santa Monica, California, aggressive pumping rates twice as great as the aquifer's natural inputs dewatered a significant portion of the upper aquifer. This, in turn, caused water containing methyl tertiary butyl ether (MTBE) to flow toward the well field from all directions and contaminate the source water (Johnson et al. 2000). This situation highlights the importance of examining pumping rates and other ways that hydrogeologic barriers can be compromised. States are encouraged to use site-specific approaches to identify hydrogeologic barriers and ensure that those barriers remain effective during the expected range of pumping operations.

Biological factors in an aquifer are considered in the definition of a hydrogeologic barrier. This refers primarily to the possibility that certain microorganisms may be predators, serving to reduce the concentrations in an aquifer of pathogens that may cause waterborne disease. Such a possibility should be evaluated on a site-specific basis because predation is highly dependent on temperature, soil type, aquifer mineralogy, and other geochemical conditions. Protistan grazing (i.e. grazing by protists) on ground water bacteria has been investigated by Kinner et al. (1998), who focused on the rates and size-selectivity of such grazing. Although the study found that up to 74 percent of the bacterial prey could be consumed fairly rapidly, this study is not directly applicable to hydrogeologic barrier investigations because the bacteria preyed upon in the study were not pathogens. Rather, these bacteria biodegraded chemical contaminants in ground water. Nasser et al. (2002) investigated the role of microbial activity in reducing concentrations of viruses in saturated soil. The results of this study, though highly dependent on virus type, are promising in terms of potential future applicability to studies of potential hydrogeologic barriers. Nevertheless, current research into the role of biological factors as potential barriers to the transport of waterborne pathogens is at a very early stage. Therefore, few, if any, PWS's will choose biological factors as hydrogeologic barriers.

The following sections present approaches for identifying hydrogeologic barriers using reliable data which may already have been collected and through field investigation. The use of site-specific field data is emphasized because hydrogeologic barriers are local phenomena, and the determination of their adequacy for protecting sensitive aquifers will often be difficult. Additional details and references regarding the types of field investigations that may be necessary are provided in Appendix A. Finally, travel time calculations may be informative if it is suspected that there is a hydrogeologic barrier consisting of a very long flow path through the unsaturated zone or through a confining unit. If the calculated travel times are longer than the expected lifetime of waterborne pathogens (see Appendix C), a hydrogeologic barrier may be present. Methods for calculating travel times, including a variety of computer models, are summarized in Appendix B.

3.5.1 Data Sources for Hydrogeologic Determinations

Many of the data sources discussed above will be useful resources for identifying potential hydrogeologic barriers. For example, source water assessment reports often indicate that a PWS well's aquifer is unconfined, in which case there will generally be no need to look further for possible confining layers. On the other hand, if documents produced by the Source Water Assessment Program (SWAP) indicate that a confining unit is present and the State wishes to pursue a potential hydrogeologic barrier investigation, EPA encourages States to use the diagnostic criteria, described below in this section, to further evaluate the ability of the unit to adequately protect the sensitive aquifer.

Well logs available through the Wellhead Protection Program (WHPP) and SWAP or directly from the driller may be the first indication that a confining unit is present. Geologic, hydrogeologic, and soil maps, if available, can be used to identify the presence, thickness, and areal extent of a confining bed. In addition, these maps may identify places and potential pathways by which fecal contamination may leak into a sensitive aquifer. Under these circumstances, the hydrogeologic barrier would not be effective in protecting the well and its associated aquifer from contamination.

Geologic maps, which depict geologic formations, may be used to determine confinement. A confining formation is commonly composed of one predominant rock type, such as shale, or of sediment, such as clay. The dominant rock or sediment type and the formation's estimated ground water production rate may indicate whether it is an effective confining unit. Again, the issue of map scale is relevant to the applicability of the information. This chapter provides information on a variety of means to obtain such maps.

Unfortunately, maps alone are unlikely to be adequate for determining whether a hydrogeologic barrier exists and is protective. The most useful data sources for hydrogeologic barrier determination will be those that describe the PWS wells and surrounding geology and hydrogeology in as much detail as possible. Again, hydrogeologic barrier determinations will generally be made on a site-by-site basis, and desktop analyses alone will often be insufficient. Given that hydrogeologic barrier determinations are not required, the detailed desktop and field investigations would be conducted at the State's discretion.

Available information from sources such as existing State, Federal, or academic hydrogeologic investigations; geologic, environmental, and hydrogeologic maps; or other data sources may be useful for identifying the presence, thickness, and areal extent of a hydrogeologic barrier. They may also identify breaches in the hydrogeologic barrier that are potential pathways for fecal contaminants to enter a sensitive aquifer. However, ensuring that a hydrogeologic barrier is functioning adequately to prevent leakage of contaminated recharge into a PWS source will very often require field data because hydrogeologic barriers are local phenomena. In most instances, some combination of site-specific field investigation and desktop analysis based on available data will be necessary.

3.5.2 Desktop Approaches

At a few sites, State, Federal, or academic institutions may have already conducted significantly detailed field investigations that show the presence of an adequate confining unit, sufficiently thick unsaturated zone, or other hydrogeologic barrier. In such instances, it may be appropriate to conclude that a hydrogeologic barrier exists, without the need to conduct additional field investigations. On the other hand, if the study in question may be outdated, it will be necessary to conduct additional research to ascertain that, for example, the confining unit has not been breached in the recent past.

4. Assessment Source Water Monitoring; Number (and Frequency) of Samples

4.1 Introduction

The GWR provides States with the option to require systems to conduct assessment source water monitoring of any ground water source at any time, and States may require systems to take corrective action based on the results of these analyses. Assessment source water monitoring allows States to initiate a more thorough source water monitoring approach than that resulting from the GWR triggered monitoring provisions on a case-by-case basis. Assessment source water monitoring also allows States to address recent development of additional source vulnerability based on sanitary surveys or other State oversight activities.

States may identify high risk GWSs and require assessment source water monitoring based on:

- Information from Source Water Assessments and sanitary surveys;
- Information from Hydrogeologic Sensitivity Assessments (HSAs);
- GWR triggered monitoring results;
- TCR monitoring results;
- Well construction information (or lack of information);
- Historical or water quality data from the system; and
- Well located in a sensitive aquifer.

The purpose of this chapter is to describe additional rationale suitable for identifying when optional assessment source water monitoring might be beneficial to protecting public health. Included in this discussion are available data to guide decisions about monitoring number and frequency, as well as information regarding sample locations, indicator choice, and analytical methods. More information on indicator choice and analytical methods can be found in the Source Water Monitoring Methods Guidance document. Corrective actions are described in more detail in the Corrective Action Guidance document.

4.2 Connection to Hydrogeologic Sensitivity Assessment

A Hydrogeologic Sensitivity Assessment (HSA) can be an effective screening tool in identifying GWSs with ground water sources that are susceptible to fecal contamination and for which assessment source water monitoring would be appropriate and beneficial. The HSA identifies wells located in sensitive aquifers that should be targeted for assessment monitoring. Chapter 3 describes the details of performing an HSA. States have the option to investigate and verify the presence of an adequate hydrogeologic barrier for sources located in hydrogeologically sensitive settings as part of a determination that assessment source water monitoring is appropriate. Section 2.5 provides more information about evaluating hydrogeologic barriers.

4.3 Assessment Monitoring Basis and Triggers

As discussed in Chapter 1, wells may be identified as candidates for assessment source water monitoring due to a variety of hydrogeologic and microbial monitoring factors. For example, assessment source water monitoring may be appropriate for wells in sensitive aquifers and wells with a history of total coliform occurrence (based on TCR monitoring). However, other factors, not hydrogeologic or microbial in nature, may also indicate wells for which assessment source water monitoring is appropriate. For example, wells located adjacent to concentrated animal feeding operations, land spreading of manure or biosolids, or near municipal landfills might be suited for such monitoring. Indicators other than microorganisms, such as MBAS, chloride, or nitrate/nitrite levels could indicate that anthropogenic activities are affecting well water quality. Where high concentrations (above normal background levels or based on historic trends) of conservative tracers (contaminants that travel at the same velocity as ground water) such as nitrate and chloride are found in well water, this indicates that there exists a relatively efficient ground water recharge pathway that permits only minor contaminant dilution and attenuation. Microbial pathogens may be more likely to arrive as infectious agents when such an efficient pathway is present. Thus, these wells could be subject to assessment source water monitoring.

4.4 Assessment Monitoring; Number (and Frequency) of Samples

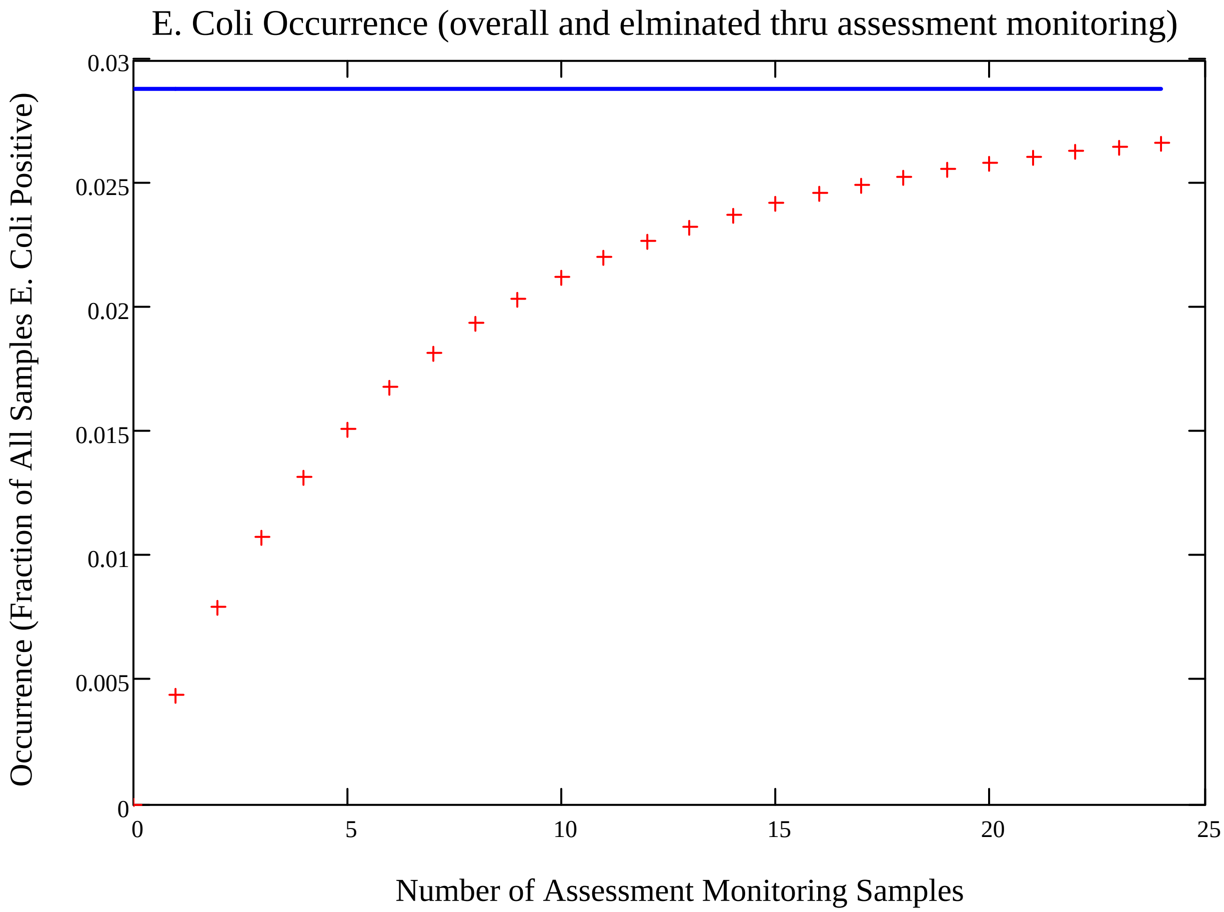
Assessment source water monitoring is one of several barriers (e.g. State sanitary setback distances, GWR sanitary surveys) designed to protect the public health of drinking water consumers using ground water sources. Ground water travel paths are complicated, and it is often difficult to establish that all ground water reaching a well emanates from a protected aquifer and has resided in the subsurface for years, decades, centuries or longer. In general, a small amount of ground water often takes the fastest path. Therefore, even in the most protected aquifer, there may be a small but significant component of recent ground water capable of carrying pathogens. As a result of this uncertainty, public health protection principles suggest that source water samples be collected as necessary. In general, public health is best protected if frequent source water samples are collected and assayed because each sample increases the probability that infrequent source water contamination, if present, is identified.

The added public health protection value due to collection of one or more source water samples is an important issue. In particular, one key aspect is identifying the additional information value of each or several source water samples. EPA evaluated this aspect using available *E. coli* data presented in the Occurrence and Monitoring background document of the Ground Water Rule (USEPA, 2006d) and in the Ground Water Rule Economic Analysis (USEPA 2006e).

Exhibit 4.1 evaluates the population of all wells randomly sampled for *E. coli*(using data from the documents described above). In Exhibit 4.1, the horizontal line at 0.029 represents the total *E. coli* proportion in untreated ground water. If all water produced by all wells could be analyzed for *E. coli* (100 ml at a time), about 2.9% of samples would be positive for *E. coli*. Because it isn't feasible to analyze all water all of the time, Exhibit 4.1 displays the proportion that would be identified if all wells were to assay the same number of samples (1, 2, ...24). The exhibit shows that less than one-half percent of the total occurrence would be identified if all wells were assayed only one sample each, but that nearly all of the occurrence would be identified if every well was assayed using 24 samples each. If each well were assayed using 5 samples, then nearly half of the wells would be identified. The exhibit shows that wells having about 80% of the *E. coli* occurrence should be identified by a positive *E. coli* result if 12 samples are assayed for each well.

Based on this analysis, twelve or more source water monitoring samples should be collected from wells identified for assessment source water monitoring, including wells in sensitive aquifers. This more frequent sampling will identify more than 80% of fecal contamination occurrences. Assessment source water monitoring should be representative of the system's typical operations. Using a minimum of 12 samples ensures sampling for each month that most systems are in operation, and addresses the impact that seasonal events can have on contamination (e.g., heavy rain events). Sampling for seasonal systems should be equally distributed (12 samples per season) or conducted during consecutive years and States may set other sampling frequencies based on local conditions.

Exhibit 4.1 Likelihood of Identifying *E. coli* Occurrence by Assessment Source Water Monitoring in a Population of Wells Randomly Selected and Sampled (Data from USEPA 2006d and USEPA 2006e)



4.5 Sample Location

Ground water samples used for triggered or assessment monitoring must be collected at a location prior to any treatment of the ground water source unless the State approves a sampling location after treatment. If the system's configuration does not allow for sampling at the well itself, the system may collect a sample at a State-approved location, provided the sample is representative of the water quality of that well.

Assessment source water monitoring sampling locations should be located as close to the wellhead as possible. For the smallest systems with the shortest length distribution systems, it might be expected that tap water and source water quality are similar; a sample at the tap might be reasonable.

Systems without sample taps at or near the well should install a sample tap

4.6 Representative Wells

The Ground Water Rule allows systems, with State approval, to identify one or more wells that are representative of multiple wells in the system for the purposes of GWR assessment source water monitoring. The representative wells are wells that draw water from the same hydrogeologic setting and that have the same, or greater, vulnerability to source water contamination as the wells that are represented (e.g., distance to source of contamination, pumping rates, etc.). Information that supports the use of representative wells includes drillers logs, well construction details, and water quality data (e.g., pH, temperature and other physical parameters, mineral analyses).

4.7 Indicator Selection

Aquifers are broadly classified into two categories; porous and non-porous. In this document, non-porous aquifers (e.g., fractured igneous or metamorphic rock aquifers) as well as gravel aquifers are defined as sensitive aquifers. Among the remaining porous aquifers (e.g., sand, or sand and gravel, aquifers), sand and gravel aquifers more efficiently transmit fecal contaminants than sand aquifers because average ground water velocity is higher.

All subsurface particles, including microbes, may be transported by flowing ground water. Particles may be removed from flow or be retarded. That is, they may permanently or temporarily become associated with the solid aquifer materials in either porous or non-porous aquifers. Microbial transport in porous media aquifers is an active research area and consensus is difficult in many issues in this field. It is generally agreed that microbe size is an important element in determining mobility in porous media, although many other factors, such as surface charge, may also have significant influence. The significant (one-thousand fold) size difference between viruses (measured in nanometers) and bacteria (measured in micrometers) increases the likelihood, other mobility factors being equal, that an infectious virus rather than an infectious bacterium, will reach a GWS well in a porous aquifer.

In other aquifers, such as non-porous aquifers (e.g., fractured igneous or metamorphic rock aquifers) and gravel aquifers, average ground water velocities are exceptionally fast, and straining and pore-size exclusion are much less significant and bacteria and viruses are assumed to travel at equal rates. In general, straining and pore-size exclusion effects are more significant in sand aquifers than in sand and gravel aquifers. In sand aquifers, ground water velocity is moderate because mean grain size is moderate. As ground water velocities increase because of increasing gravel content or increasing proximity to a pumping well, the differences between virus and bacterial transport efficiency become less important, and either a viral or bacterial indicator may be recommended (See Exhibit 1.1).

On the other hand, the finest grained porous aquifers, such as shale and clay beds, are not considered to be aquifers because ground water velocities through them are generally very slow. Thus, despite the great significance of straining and pore-size exclusion in such environments, entrained pathogens are not transmitted efficiently through shale or clay, therefore such subsurface formations are not considered further in this guidance.

4.8 Analytical Methods

The Ground Water Rule Source Water Monitoring Methods Guidance document provides a detailed explanation of EPA-approved analytical methods. A compilation of these methods is listed in Exhibits 4.2, 4.3, and 4.4. Details about each method can be found in the Source Water Monitoring Methods Guidance document. The last column in each table identifies the section of the Source Water Monitoring Methods Guidance document where additional explanatory text resides. The tables list approved methods for GWR monitoring as of the date of this guidance document. Additional methods may have been approved for GWR monitoring since this date. Additional information, including additional approved analytical methods for GWR monitoring, can be found at the following URL: http://www.epa.gov/safewater/methods/methods.html.

Exhibit 4.2 *E. coli* Methods Approved for Use under the GWR

Media	Method Reference	Approved Formats	Description of Positive Result	Section
Colilert®	SM[1] 9223	Presence/Absence Multiple-Well Multiple-Tube	Yellow, fluorescent	6.2.1
Colisure®	SM[1] 9223	Presence/Absence Multiple-Well Multiple-Tube	Red/magenta, fluorescent	6.2.2
E*Colite	—	Presence/Absence	Blue/green, fluorescent	6.2.3
LTB6 EC-MUG	SM[1] 9221B6 SM[1] 9221F	Presence/Absence Multiple-Tube	Growth and the presence of acid and/or gas in LTB, fluorescent in EC-MUG	6.2.4
mEndo or LES Endo6 NA-MUG	SM[1] 9222B6 SM[1] 9222G	Membrane Filtration	Pink to red colonies with metallic (golden-green) sheen that fluoresce after transfer to NA-MUG	6.2.5
MI Medium	EPA Method 1604	Membrane Filtration	Blue colonies	6.2.6
m-ColiBlue24®	—	Membrane Filtration	Blue colonies	6.2.7

[1]*Standard Methods for the Examination of Water and Wastewater*, 20th edition.

Exhibit 4.3 Enterococci Methods Approved for Use under the GWR

Media	Method Reference	Approved Formats	Description of Positive Results	Section
Azide Dextrose / BEA / BHI	SM[1] 9230B	Presence/Absence Multiple-Tube	Growth at 45EC in BHI and growth in BHI with 6.5% NaCl at 35EC	6.3.1
mE-EIA	SM[1] 9230C	Membrane Filtration	Pink to red colonies that form black or reddish-brown precipitate on underside of filter	6.3.2
mEI	EPA Method 1600	Membrane Filtration	All colonies with a blue halo	6.3.3
Enterolert™	D6503-99[2]	Presence/Absence Multiple-Well Multiple-Tube	Presence of blue-white fluorescence	6.3.4

[1]*Standard Methods for the Examination of Water and Wastewater*, 20th edition.

[2]Annual Book of ASTM Standards Water and Environmental Technology, Volume 11.02, 2000.

Exhibit 4.4 Coliphage Methods Approved for Use under the GWR

Media	Method Reference	Approved Formats	Description of Positive Result	Section
Two-Step Enrichment	EPA Method 1601	Presence/Absence	Presence of plaques (circular lysis zones)	6.4.1
Single Agar Layer	EPA Method 1602	Presence/Absence Quantitative	Presence of plaques (circular lysis zones)	6.4.2

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Appendix A: Field Methods for Determining the Presence of a Hydrogeologic Barrier

Pump and Slug Tests

Pump tests can be used to estimate aquifer properties on local and regional scales. A pump test requires pumping water at a known rate from a test well or a production well, and measuring water-level drawdown over time in the pumping well and/or nearby monitoring wells (Witten and Horsley, 1995).

Careful test design is critical to obtaining accurate pump test data. A monitoring well can be sited by estimating the pumping rate of the well and the expected drawdown. It is important to locate monitoring well screens in the same hydrostratigraphic unit as the screen of the pumped well (Witten and Horsley, 1995). Driscoll (1986) and Walton (1989) provide further information on pump test design. Pump test analytical methods are available for unconfined, confined, and semi-confined (leaky) aquifers. Pump tests are usually analyzed using the drawdown data collected during pumping, but tests may also use the hydraulic head recovery measurements taken as the potentiometric surface returns to its initial level after pumping is ceased (Witten and Horsley, 1995).

Slug tests may also be useful to determine aquifer properties such as hydraulic conductivity and transmissivity. When a slug (a solid rod, or, alternatively, a given volume of additional water) is dropped into a well, the water level in the well is forced to suddenly rise, and the rate at which as the water level returns to its initial level can be measured. The initial test results can be verified by running the test in reverse (i.e., by removing the slug from the well). To achieve reliable test results, it is important that a well be properly constructed (Witten and Horsley, 1995).

Geophysical Methods

Borehole geophysical data can be used to help interpret which fractured bedrock intervals have low permeability and may act as confining layers. These well logging devices are usually run in combination and can include caliper, resistivity, spontaneous potential, neutron, gamma, and television logs. Risser and Barton (1995) report that caliper, single-point resistance, and gamma logs are commonly used for identifying fractures and fracture zones. Television logging is particularly useful for identifying vertical fracturing. Comparatively expensive, unconventional logs, such as full-waveform acoustic, acoustic televiewer, vertical seismic profiling, borehole radar, and resistivity tomography may provide good data on fractured bedrock aquifers.

Borehole flowmeters are designed to measure the flow into or within a well bore. Heat pulse, electromagnetic, and impeller flow meters are designed to locate productive fracture zones (Molz *et* al. 1990). If no productive fracture zones are identified in the upper part of the aquifer, the interval may act as a confining layer.

Ground Water Age Dating

Ground water age dating of samples collected from a public water supply well can help quantify ground water travel time from the surface to the well. At locations where ground water travel times are short, ground water velocities are greater for a given flow path length (well depth). For the short ground water travel times characteristic of sensitive aquifers, helium-3/tritium ratios, and oxygen and hydrogen isotope concentrations are best suited for age dating. Although these methods are being developed in a research setting, their application to sensitive aquifers has not been routine. Risser and Barton (1995) caution that age dating can lead to erroneous conclusions if a well is receiving water from two or more distinct intervals with significantly different ground water ages. The mixed water will provide ground water ages intermediate between the two actual values.

Tritium, the radioactive isotope of hydrogen with a half-life of 12.38 years, was released from nuclear bomb tests in the atmosphere and serves as a time marker. For ground water with travel times of a few months, the helium-3/tritium ratio can be used (Beyerle *et* al. 1999; Stute *et* al. 1997). The concentration of helium-3 dissolved in ground water increases as soon as the ground water is isolated from the atmosphere because helium-3, which is produced by tritium decay, can no longer escape. Therefore, the helium-3/tritium ratio is a measure of the time elapsed since a water parcel was last in contact with the atmosphere (Beyerle *et* al. 1999). Other sources of radiogenic helium may also be present, produced by the decay of uranium and thorium in mineral grains; care is needed in using this technique (Solomon *et* al. 1992).

Stable isotope ratios such as oxygen-18/oxygen-16 and deuterium/hydrogen can also be used to identify the young ground water (i.e., ground water with short residence times) typical of sensitive aquifers. Stable isotope methods are designed to compare the isotopic character of different waters by using plots of isotope ratios and their deviation from a recognized standard. Because isotopic ratios differ by season, seasonal recharge can sometimes be recognized in ground water. With a sufficient surface water stable isotope record, short residence periods in ground water can be determined by their differing isotopic signatures (Beyerle *et* al. 1999).

Tracer Testing

The most direct way to determine ground water velocity is by introducing a tracer substance at one point in the flow field and observing its arrival at another point in the flow field (typically a monitoring well) (Freeze and Cherry, 1979). Tracer tests are the most conclusive method for evaluating the direction and travel time of ground water flow in bedrock aquifers (Risser and Barton, 1995). Tracers can consist of organic dyes, inorganic salts, gas, or solid particles. Tracer recovery at a given location indicates a hydraulic connection, and the time required to detect the tracer can be used to calculate ground water velocities. Additional information on tracer testing can be obtained from Aley and Fletcher (1976) or Field (1999).

Appendix B: Ground Water Travel Time

Introduction

Ground water travel time (GWTT) is the time that it takes a small amount or “packet” of ground water to move from one point in the aquifer to an endpoint (e.g., a pumping well). It is sometimes helpful to use a circle or ellipse on a map to represent the area surrounding a continuously pumping well that will contribute water to the well after, for example, 2 years of travel. The enclosed area is sometimes referred to, for simplicity, as the “travel time zone.” Such zones can be delineated and drawn on a map when the ground water travel times are known from many locations in the aquifer, and an appropriate travel time of interest is chosen. In order for such maps to be meaningful, it is necessary to presume horizontal flow conditions. This can be a useful method for representing areas on a map that may contribute to the contamination of a pumping well over the time period of interest.

Pumping increases the rate of ground water flow and shortens the time it takes for ground water (and any associated contaminants and pathogens) to move from one point in the aquifer to another. In sand, sandstone, and similar aquifer types, ground water travel time can be calculated using previously determined estimates of flow system parameters, including hydraulic conductivity, induced hydraulic gradient, and porosity. Alternatively, travel time estimates can be based on measurements of natural or artificial tracer transport. Tracers mimic the behavior of ground water itself, and ideally have little chemical interaction with aquifer material. Due to natural spreading or “dispersion” in the subsurface, a certain amount of any tracer and the ground water which carries it has an arrival time at a given well that is shorter than the average arrival time. This is the result of some water (and tracer) molecules taking a more direct path rather than the more typical tortuous path through the granular aquifer. In general, viruses and bacteria transported through ground water typically arrive later than the average water “packet.” However, due in part to their small size, some viruses may take the fastest path from the source to a well and arrive before the average ground water travel time. Thus, the United States Environmental Protection Agency (EPA) recommends that estimates of ground water travel times be interpreted carefully in order to ensure that proposed hydrogeologic barriers are truly protective.

Because viruses are acute contaminants, capable of causing infection at very low doses, one possible strategy to protect the public is to focus on those viruses that arrive at wells in an infective state. If a particular virus may survive for four to six months in the subsurface, it may be advisable to add another six-month “safety factor” to the determined ground water travel time to account for uncertainties such as dispersion in the calculation of travel times.

Ground water travel time calculations often necessitate the use of computer models. All methods for calculating GWTT require simplifying assumptions. Even the most complex ground water calculations (three-dimensional numerical ground water flow models) are simplified representations of the aquifer/well system. EPA (1987b) has grouped the GWTT calculation methods into four groups. These groups are (in order of increasing computational complexity): (1) uniform flow, (2) analytical, (3) semi-analytical, and (4) numerical methods. Under certain conditions, described below, an even simpler one-dimensional method may be used.

All methods considered in this document are based on the simplifying assumption of steady state ground water flow. Under the steady state assumption, the well pumping effects do not change with time and ground water flow has achieved a new equilibrium (different than the natural equilibrium prior to the start of pumping). When the actual well pumping rate is variable over the time period of interest (several months or 1 or 2 years), for the purposes of evaluating a possible hydrogeologic barrier, the well is assumed to be uniformly pumped at the maximum sustainable rate for the entire time period of interest.

The simplifying assumption of steady state flow is a technical requirement necessary to perform uniform or analytical calculations. However, it is possible to perform semianalytical or numerical GWTT calculations without this simplifying assumption. Nevertheless, by applying the steady state assumption to all methods discussed here, comparisons among the methods can more easily be performed.

All GWTT calculation methods require input of the average effective porosity of the aquifer. Porosity, in a saturated portion of an aquifer, is the proportion of interparticle void space that is filled with water. Void space varies within an aquifer, so the porosity value is averaged to simplify GWTT calculations. *Effective* porosity refers to that portion of an aquifer's void space through which water can travel. This definition is necessary because some water is trapped in pores that are sealed in all directions by mineral growth. Other water is bound in very small pores in clay minerals and is capable of moving only over time scales longer than those of concern here. The term, "porosity," as used in the following discussion, refers to average, effective porosity.

Commonly, porosity measurements in the vicinity of a well are unavailable, even within the largest aquifers. More typically, porosity values are estimated, based on knowledge of the aquifer's hydrogeologic setting (e.g., alluvial) or, for regional extensive aquifers, based on the aquifer type known from the name of the hydrogeologic unit (e.g., Dakota sandstone). Porosity values typical of alluvial or sandstone aquifers are available in the scientific literature - for example, in Freeze and Cherry (1979). Although porosity values can vary by 10 or 20 percent even within a given aquifer category, variations in hydraulic conductivity are typically several orders of magnitude. Thus, heterogeneity of hydraulic conductivity is usually the largest single factor introducing uncertainty into ground water flow models and GWTT calculations which are not based on tracer tests.

One Dimensional Method for Horizontal Flow

The one dimensional (1D) GWTT method is a simple equation for calculating horizontal time of travel (USEPA 1994a). The method is most appropriate for natural ground water flow (i.e., when there is no pumping well) in localities where data are available for three input parameters that describe the properties of the aquifer: horizontal hydraulic conductivity, horizontal hydraulic gradient, and porosity. EPA (1991b) suggests a supplementary approximation method that can be used to simulate GWTT in areas of steep water table slope, such as in the vicinity of a pumping well. Nevertheless, because the 1-D GWTT method does not explicitly account for the presence and actions of a pumping well, it is generally not appropriate for source water assessment or hydrogeologic sensitivity assessment purposes. The mathematical expression is presented in EPA (1994a, p. 74).

Calculated fixed radius GWTT method

The calculated fixed radius GWTT method explicitly accounts for the presence and actions of a pumping well. Thus, it is appropriate for GWTT calculations for Source Water Assessment Program (SWAP) or Ground Water Rule (GWR) purposes. The method is based on an assumption of cylindrical flow to a well with pore volume equal to the pumped volume of water during the specified period. Everywhere in the cylinder, water flows horizontally to the well.

The calculated fixed radius GWTT method is a simple calculation that requires only three input parameters. Furthermore, of the three input parameters, only the porosity is a property of the aquifer; the other two parameters describe the construction or operation of the well. In general, it is easier to obtain site-specific values for the latter parameters. As discussed above, porosity values are typically estimated rather than measured. The required well parameters are the well pumping rate and the length of the open interval or well screen. The mathematical expression of the calculated fixed radius GWTT method is presented in EPA (1994a, p. 70).

The calculated fixed radius GWTT method is valid if the drawdown from pumping is less than about 10 percent of the aquifer's pre-pumping saturated thickness (Reilly *et* al. 1987). Furthermore, the method requires that the well fully penetrate the water-bearing zone of the aquifer and be open or screened throughout the entire interval. The method is most appropriate for aquifers that most closely approximate a homogeneous, isotropic aquifer of constant thickness located in a region with a flat water table. Hydrologic boundaries are assumed to be sufficiently distant so that the ground water flow field in the vicinity of the well is not significantly affected by those boundaries. It is assumed that all flow is horizontal and ground water flow velocity is constant. The method can be modified to simulate constant flux recharge or leakage (Risser and Madden, 1994). Risser and Madden (1994) also investigate the effects of violating the flat water table assumption.

EPA suggests that the method be applied only to shallow coastal plain, wide topographic basin, or mid-continent aquifers where the aquifer and the topography are relatively flat. Risser and Madden (1994) suggest that the method is not appropriate for the valley-fill aquifers in Pennsylvania. Similarly, EPA suggests that the method is not appropriate for any dimensionally restricted aquifer (i.e., those that are long and narrow such as a rift-basin, barrier island, glacial buried valley, alluvial, or esker aquifer). The method is not appropriate for wells that are located near surface water or near a topographic high point or in areas with irrigation wells or other high demand ground water usage.

Uniform Flow GWTT Method

The uniform flow method is an analytical solution that can be used to estimate GWTT for steady flow to a well. The uniform flow equations are derived by superposition of the Dupuit equation for radial flow around a well with the one-dimensional, uniform, pre-pumping flow field (Risser and Madden 1994).

The method requires six input parameters. Five parameters describe the properties of the aquifer; porosity, hydraulic conductivity, aquifer thickness, and hydraulic gradient magnitude and direction. The remaining parameter describes the pumping well pumping rate.

To use the uniform flow method, the aquifer is assumed to be confined, of constant thickness, homogeneous, and isotropic. The pre-pumping potentiometric surface may be flat or uniformly sloping and pumping from a fully-penetrating well is assumed to have resulted in a steady state. According to Risser and Madden (1994), the assumptions of a steady-state flow and a uniformly sloping potentiometric surface are not theoretically possible in an unbounded aquifer; but, if boundaries are distant, the assumptions may be valid. The method may be used for unconfined aquifers if the well drawdown is less than about ten percent of the pre-pumping saturated thickness of the aquifer. Mathematical expressions of uniform flow are presented in EPA (1994a, p. 81) and Risser and Madden (1994, p. 32) The most efficient approach to make use of the uniform flow method is to use the wellhead protection area (WHPA) model (USEPA 1993b), which has the added benefit of being applicable to a wider variety of aquifer boundary types without requiring more input parameters describing the aquifer or well properties.

Similar to the calculated fixed radius method, the uniform flow method is applicable to shallow coastal plain, wide topographic basin or mid-continent aquifers where the aquifer and the topography are relatively flat. EPA suggests that States find multiple solutions in order to bound the uncertainty in the estimated parameters used with this method.

Analytical and Semi-Analytical GWTT Methods

Analytical and semi-analytical methods comprise a group of equations that are superimposed (equation solutions are added together) in various combinations to simulate particular aquifer settings and types. In the more simple combinations, the method does not require any additional knowledge about aquifer or well properties beyond that required for the uniform flow equation. Additional parameters needed are used to define the aquifer boundaries (i.e., the distance to the nearest significant surface water body or rock outcrop that forms a barrier to ground water flow). Typically, combinations of surface water or rock outcrops can be used to bound the aquifer on two or four sides.

Because of the computational complexity of the analytical methods, semi-analytical methods are typically used. The semi-analytical method makes use of more advanced computational methods with the help of a computer, but does not change the number of significant input parameters. The WHPA model (USEPA 1993b) is one example of a semi-analytical method that solves the complex analytical flow equations. The user-interface for the WHPA model shields the user from the complexity of the method.

The WHPA model is divided into two basic modules: one that is suitable for an isolated well and one that is suitable for a well in a well field where nearby wells have a significant interfering effect on the flow to each well. The module MWCAP (Multiple Well Capture Zone) (or RESSQC) is suitable for the former but not the latter. The module General Particle Tracking (GPTRAC) is suitable for either case.

The minimum input parameters are the same for both MWCAP and GPTRAC. The input parameters are the same six parameters as those required by the uniform flow equation. Five parameters describe the properties of the aquifer: porosity, hydraulic conductivity, aquifer thickness, hydraulic gradient magnitude and direction, and the sixth parameter, well pumping rate, describes the pumping well.

The assumptions governing the use of the analytical and semi-analytical methods are similar to the uniform flow equation in terms of the properties of the aquifer, but they are dissimilar in terms of the boundaries of the aquifer. To use the analytical and semi-analytical methods, the aquifer is assumed to be confined, of constant thickness, homogeneous and isotropic. The prepumping potentiometric surface may be flat or uniformly sloping, and pumping from a fully-penetrating well is assumed to have resulted in a steady state condition. The method may be used for unconfined aquifers if the well drawdown is less than about 10 percent of the pre-pumping saturated thickness of the aquifer. Unlike the uniform flow equation assumptions, the analytical method may be used near surface water or near a topographic high. If GPTRAC is used, as discussed above, the semi-analytical method may be used to calculate GWTT to a well adjacent to another, interfering, well.

The GPTRAC module of WHPA may also be used to calculate GWTT in a leaky, confined aquifer. However, three additional input parameters that describe the properties of the aquifer are required; confining bed hydraulic conductivity, confining bed thickness, and areal aquifer recharge rate. The module GPTRAC may also be used to calculate GWTT in an unconfined aquifer (if drawdown is less than 10 percent of the aquifer thickness) where areal aquifer recharge is significant. In this application, information is needed to specify a boundary condition. This boundary condition is the distance at which the well pumping effects are negligible. Such information is typically not available, thus necessitating that this application be used with caution. Also, for this application, the areal recharge rate is needed.

EPA suggests that the analytical and semi-analytical methods are applicable to all granular, porous aquifers. Hansen (1991) suggests that the analytical and numerical flow methods could be used for calculating GWTT (for microbial protection areas) around public water supply wells near Mt. Hope, Kansas. These wells are located in an unconfined, unconsolidated, Quaternerary terrace and alluvial deposits of silt, clay, and gravel (High Plains aquifer). Risser and Madden (1994) report that the semi-analytical method is a powerful and flexible method that may be used as long as the water table surface is not highly irregular in shape. Lerner (1992) suggests that the semi-analytical ground water flow model ROSE performs better at calculating GWTT for one type of aquifer setting (i.e., aquifers with significant recharge and with a well distant from the impermeable boundary). For hydrogeologic settings in which the well only partially penetrates the full saturated thickness of the aquifer, an analytical solution is available (Faybishenko et al. 1995).

Analytical Element GWTT method

The analytic element method is the most recent development of solution techniques for ground water flow (Strack 1989; Haitjema 1995). The analytical element method uses both analytical and numerical methods to perform the ground water travel time calculation. The EPA analytical element method, wellhead analytical element model (WHAEM) (USEPA 1994a; Kelson et al. 1999, Kraemer et al. 1999) is designed to calculate GWTTs. To perform the GWTT calculation, WHAEM requires the following parameters: porosity, hydraulic conductivity, aquifer thickness, areal recharge rate, pumping rate, and stream water levels. Because it uses a more sophisticated computational method than WHPA, WHAEM is also capable of simulating hydrogeologic settings in which streams are not fully incised through the entire saturated thickness of the aquifer (partially penetrating streams). Both WHPA and

WHAEM can simulate large water bodies that perform as hydrogeologic boundaries because they fully penetrate the aquifer. However, unlike WHPA, WHAEM can also simulate partially penetrating streams that do not perform as hydrogeologic boundaries and that gain and lose ground water independent of areal recharge and well pumping.

WHAEM is capable of simulating any area around a well that is shaped as a polygon. WHAEM can also simulate uniform stream or barrier boundaries on each side of the rectangle or polygon.

WHAEM is applicable to all hydrogeologic settings with porous media (e.g., sand, gravel). If the hydrogeologic setting appears complex, such as at the confluence of two large rivers or at the confluence of a large and small river, then WHAEM may provide a more accurate GWTT calculation than WHPA.

Numerical GWTT method

Numerical methods have the capability to simulate nonlinear, nonfully penetrating boundary conditions, complex patterns of recharge and discharge, and spatial heterogeneity of hydraulic properties (Risser and Madden, 1994). These methods require, at a minimum, all of the input parameter data required by semi-analytical methods. More complex simulations require substantial amounts of input data and are not further considered here.

Calculated GWTT for Vertical Ground Water Flow in Granular Porous Aquifers

The vertical ground water flow to be discussed here is natural gradient flow directed vertically downward in saturated porous media. The flow is assumed to be far from a pumping well so that the drawdown due to well pumping has no effect on the vertical GWTT. An unsaturated and saturated glacial till overlies a karst limestone aquifer in Minnesota. The unsaturated portion and a saturated portion of glacial till is believed (by Minnesota drinking water staff) to act as a hydrogeologic barrier that prevents fecal contamination of the karst limestone aquifer. The vertical GWTT is the travel time for vertical ground water flow through the saturated glacial till component of the hydrogeologic barrier.

The flow through the unsaturated glacial till component is also part of the hydrogeologic barrier. However, unsaturated ground water flow is much more complex than saturated ground water flow because the hydraulic conductivity cannot be assumed to take the same value everywhere in the medium. Rather, in unsaturated soil and aquifer material, the hydraulic conductivity is very strongly dependent on the porous medium percent saturation. The percent saturation is typically low (20-30 percent) near the ground surface and increases to 100 percent at the water table. At a particular saturation point, the unsaturated material may have two hydraulic conductivity values, depending on whether the material is in a wetting phase from recent infiltrating precipitation or a drying phase as drainage, evapotranspiration, and recharge remove water from the material. Unsaturated drainage may flow in finger-like wetting fronts rather than as a uniform drainage front. Some fingers may drain very quickly compared to the overall movement of the drainage front. As a result of these physical processes, unsaturated flow is very complex, and vertical GWTT calculations are difficult.

For most aquifers covered by a thin layer of unsaturated materials (a few tens of feet is typical of the humid eastern United States), including the glacial till in the above example, it might be appropriate to assume that the entire aquifer thickness is saturated. That is, saturated conditions begin at the ground surface, as might be true after a heavy rain. This assumption avoids the complexity associated with unsaturated flow.

Darcy's Law GWTT Method for Vertical Flow Between a Confined Aquifer and an Unconfined Aquifer

The calculation of GWTT using Darcy's Law for vertical flow to or from a confined aquifer is similar to the Darcy's Law calculation for horizontal GWTT (USEPA 1991b, 1994). Input parameters needed are: 1) water level (hydraulic head) for the confined aquifer, 2) water level at the water table, 3) vertical hydraulic conductivity of the confining layer, 4) confining layer thickness, and 5) confining layer porosity.

Vertical GWTT Method for Flow from the Water Table to the Bottom of an Unconfined Aquifer

The vertical GWTT for flow from the water table to the bottom of an unconfined aquifer may be difficult to determine. This is because data from nests of piezometers, which measure hydraulic head at more than one vertical location in an aquifer, are often unavailable. Vertical GWTT determinations also require knowledge of vertical hydraulic conductivity (assuming the aquifer is anisotropic, as most aquifers are), porosity, and the thickness of the unconfined aquifers, which varies with any variation in the depth of the water table. Seasonal recharge, barometric pressure, and heavy pumping can have significant effects on the depth of the water table. Assuming all data inputs could be accurately determined, vertical GWTT's could be calculated using Darcy's Law.

Vertical GWTT Method for Unsaturated Flow to the Water Table

One-dimensional vertical downward flow through the unsaturated zone to the water table is a component of the Virus Analytical Transport (VIRALT) and composite analytical-numerical model for viral and solute transport simulation (CANVAS) methods (Hydrogeologic, Inc., 1994; 1995). Both methods use the same GWTT calculation method, which is based on the finite element and semi-analytical code for simulating one-dimensional flow and solute transport in the unsaturated zone (FECTUZ) numerical and semi-analytical, unsaturated GWTT method (Hydrogeologic, Inc., 1988).

The VIRALT/CANVAS method includes a built-in database of input parameter values from 12 typical soils that range from predominantly clay-rich to predominantly sand-rich along with various combinations of sand, silt, and clay-sized particles. The method allows the user to select one of the 12 soil types for each unsaturated horizon; when selected, the database provides most of the needed input parameter values for that soil. Alternatively, the user can specify each input parameter separately based on site-specific data.

The flow of water in the unsaturated zone is assumed to be vertically downward (one-dimensional). The flow is also considered to be at steady-state, isothermal, and governed by Darcy's Law. The aquifer is assumed to be homogeneous, the ground water slightly compressible, and the effects of wetting and drying cycles on the ground water flow parameters are neglected. Recharge rates may vary in time, but the flow field is assumed to adjust instantaneously from an existing steady-state condition to one reflective of the new recharge rate. Up to 10 unsaturated layers with differing properties may be specified, but each layer is assumed to be a uniform and incompressible porous medium (Hydrogeologic Inc., 1994).

For each unsaturated interval, the following input parameters are needed if the built-in soil database is not used: 1) saturated hydraulic conductivity, 2) saturated water content, 3) residual water content, 4) empirical parameter alpha, 5) empirical parameter beta, and 6) infiltration rate. The method is applicable to all unsaturated porous media.

Appendix C: Microbial Inactivation Rates

Microorganisms can enter aquifers due to failed septic systems, faulty construction of waste disposal injection wells, leaking sewer lines, infiltration from surface water impoundments, or ground water interaction with contaminated surface water bodies (Bedient et al. 1999). Once released, the microorganisms are faced with stresses imposed by the environment, including competition with other microorganisms. Some released bacteria can replicate in the natural environment, but parasitic protozoa and viral pathogens are unable to replicate in natural settings.

The United States Environmental Protection Agency (EPA) is interested in predicting the fate of those microorganisms released into the environment as a consequence of human activities. The Ground Water Rule (GWR) is based on estimates of the length of time over which bacterial and viral pathogens might pose a hazard to public water supply (PWS) wells. After a certain amount of time, a released pathogen will no longer pose a human health hazard due to inactivation.

According to Hurst (1997), microorganisms released into the environment become susceptible to inactivation by a variety of physical, chemical, and biological processes. These processes include desiccation, denaturation, biochemical antagonism from enzymes, and predation. The inactivation rates may be accelerated by temperature, pH, interaction with inorganic and organic dissolved and solid phases, and solar radiation in surface environments (Yates and Yates 1988).

Microbial survival studies are designed to evaluate the time during which microorganisms remain viable. Hurst (1997) reviews the variety of methods, procedures and objectives for performing such studies. Microbes are generally too small to be monitored as individual organisms. Rather, population survival is studied. Hurst believes that it is best to study microbial populations within their natural environment, where they are free to move and exchange chemicals with their surroundings. However, such studies for large microbial populations are unlikely to be practical for the purposes of a hydrogeologic sensitivity assessment (HSA). As an alternative, microorganism survival can be studied in containers placed within the natural environment, such as in a well, or in environmental media (e.g., ground water samples) brought into the laboratory.

Potential pathogen hazard to PWS wells can be evaluated by identifying the longest period before inactivation occurs for one or more pathogenic bacteria or viruses. Given knowledge of the length of the potential hazard period, States can identify wells that may be sensitive to pathogen contamination due to the short travel time for ground water recharge (potentially containing pathogens) to that well.

The microbial survival data in this review are restricted to studies of fecally-derived bacteria and viruses that are most commonly transmitted via oral ingestion of drinking water. Pathogenic protozoa, such as *Giardia* and *Cryptosporidium*, if found in ground water-supplied PWS systems, would often result in those PWS's being classified as GWUDI and subject to the requirements of the Surface Water Treatment Rule rather than the GWR. Thus, the survival of protozoa will not be discussed here. Virus survival in ground water and surface water were compared by Hurst (1998), who concluded that statistical models used to predict virus inactivation in surface water could not be applied to ground water. The reasons for these differing inactivation characteristics are not known, but they could be due simply to the ubiquity of naturally antagonistic microorganisms in surface water.

Microbial survival studies can be conducted in ground water by using a very large number of amendments, including nutrients, soil, aquifer materials, wastewater, detergents, and waste products. Each of the amendments has a potential confounding effect on microbial survival. In order to minimize the confounding effects, only survival studies that were conducted in unamended ground water (studied either *in situ* or in the laboratory) are included in the table below. Furthermore, the table lists only studies of unamended ground water conducted at temperatures within the range of 8 to 25 degrees C. Such temperatures are typical of ground water in the United States, measured at depths of approximately 30 meters (m).

A survey of the scientific literature identified numerous bacterial and virus inactivation studies conducted in unamended ground water at typical ground water temperatures. These studies are listed in the following table and show the longest survival periods (lowest inactivation rates) for various bacterial and viral pathogens and fecal indicator organisms. An inactivation rate of 0.1 can be interpolated to indicate four log microbial inactivation in 40 days. Similarly, an inactivation rate of 0.01 indicates 4-log microbial inactivation in 400 days. The longest survival rate for pathogenic viruses is typically about 0.02, which indicates 4-log microbial inactivation in approximately 200 days

The virus and bacterial inactivation rate data below are unlikely to be accurate when antagonistic microorganisms are present in the ground water sample studied. In some studies, ground water samples were sterilized or filtered before the test population was seeded into the sample. In other studies, no sample treatment occurred. Any sample treatment is noted in the table, and the collection location of the ground water sample is provided if available. In one study (Biziagos *et* al. 1988), mineral water (ground water collected from a spring) was sampled and because the data met the criteria of being unamended ground water at a typical ground water temperature, the information was included in the table. The mineral water may, however, be atypical.

Greatest Survival: Laboratory-Measured Virus or Bacterial Inactivation Rates in Pure Ground Water at Ground Water Temperatures (8-25 degrees C)

Reference	**Pathogen/ Indicator**	**Inactivation rate (log_{10}/day)**	**Temp. (deg. C)**	**Sterile/ filtered**	**Hydrogeologic Setting**
Bacteria					
McFeters (1974)	*Shigella dysentariae*	0.74	9-12.5		
McFeters (1974)	*Shigella sonnei*	0.68	9-12.5		
McFeters (1974)	*Shigella flexeri*	0.62	9-12.5		
Rice et al. (1992)	*E. coli* O157:H7	0.1428	20		
Keswick (1982)	fecal streptococcus	0.23	3-15		
Keswick (1982)	*Salmonella typhimurium*	0.22	12-20		
Keswick (1982)	fecal coliform	0.36	12-20		
Keswick (1982)	*E. coli*	0.32	12-20		
Nasser (1999)	*E. coli*	0.019 (est)	20		
McFeters (1974)	*Vibrio cholerae*	2.31	9-12.5		
Viruses					
Bitton et al. (1983)	Poliovirus 1	0.02	24		
Biziagos (1988)	Poliovirus 1	0.0193	23	filtered (bottled mineral water)	Puy de Dome Spring Auvergne, France
Nasser (1999)	Poliovirus 1	0.011 (est.)	10		
Yates (1992)	Echovirus 1	0.02702	12	filtered	

Reference	**Pathogen/ Indicator**	**Inactivation rate (log_{10}/day)**	**Temp. (deg. C)**	**Sterile/ filtered**	**Hydrogeologic Setting**
Yates (1992)	Male-specific coliphage (MS-2)	0.02841	12	non-filtered	
Nasser (1999)	F+Phage	0.011 (est.)	10		
Jansons (1989)	Coxsackievirus B5	0.05	19.4		
Sobsey (1986)	Hepatitis A Virus	0.0357	25		
Biziagos (1988)	Hepatitis A Virus	0.0166	23	filtered (bottled mineral water)	Puy de Dome Spring Auvergne, France
Nasser (1999)	Hepatitis A Virus	0.00 (est.) 0.18 (est.)	10 20		
Pancorbo (1987)	Rotavirus	0.158	20		
Grondin (1987)	f2	0.1	20		
Gerba (Undated)	Simian Rotavirus	0.11968	23		
Yahya (1993)	PRD-1	0.1 (est)	23		

Appendix D: Additional Reference Sources

Appendix A of EPA's 1994 *Ground Water and Wellhead Protection Handbook* contains an extensive list of additional general reference sources that can be consulted for further information. Some of the most relevant references from that Handbook are included in the list below. In addition, several more current general reference sources are listed. The *Ground Water and Wellhead Protection Handbook* can be found online at: (http://yosemite.epa.gov/water/owrccatalog.nsf/9da204a4b4406ef885256ae0007a79c7/e05465d7e89be57d85256b0600723c19!OpenDocument.)

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