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# TECHNICAL PROTOCOL FOR IMPLEMENTING THE GROUNDWATER CIRCULATING WELL TECHNOLOGY FOR SITE REMEDIATION

**Battelle**

**December 9, 1997**

ESTCP

ENVIRONMENTAL SECURITY TECHNOLOGY CERTIFICATION PROGRAM

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December 10, 1997

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Attention: Mr. Terry Messenger

Dear Mr. Messenger

Enclosed please find 1 copy of the draft Final Report entitled "Technical Protocol for Implementing the Groundwater Circulation Technology for Site Remediation." The comments from the various reviewers were taken into account and the protocol now follows the approved outline. Please review the document and forward any comments to me as soon as possible. I will incorporate any necessary changes to produce the final version. If you have any questions about the protocol, please call at (614) 424-5715.

Sincerely,

Bruce C. Alleman

Bruce C. Alleman, Ph.D.
Research Leader
Environmental Restoration Department

BCA:rsc
Enclosure

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| | |
|---|---|
| 3-D | three dimensional |
| bgs | below ground surface |
| BOD | biological oxygen demand |
| BTEX | benzene, toluene, ethylbenzene, and xylenes |
| DCE | dichloroethylene |
| DDC | density driven convection |
| DNAPL | dense, nonaqueous-phase liquid |
| DO | dissolved oxygen |
| EPA | Environmental Protection Agency |
| GAC | granular activated carbon |
| GC | gas chromatograph |
| GC/MS | gas chromatograph/mass spectrometer |
| GCW | groundwater circulating well |
| GMS | groundwater modeling system |
| HRT | hydraulic residence time |
| ICE | internal combustion engine |
| IDW | investigation-derived waste |
| $K$ | hydraulic conductivity |
| $k$ | permeability |
| Kh | horizontal hydraulic conductivity |
| Kv | vertical hydraulic conductivity |
| LEL | lower explosive limit |
| LNAPL | light, nonaqueous-phase liquid |
| MEK | methyl ethyl ketone |
| MIBK | methyl isobutyl ketone |
| MMOC | modified method of characteristics |
| MOC | method of characteristics |
| OD | outside diameter |

| | |
|---|---|
| ORP | oxidation/reduction potential |
| PAH | polycyclic aromatic hydrocarbon |
| PCB | polycyclic biphenyl |
| PCE | perchloroethylene |
| PVC | polyvinyl chloride |
| RCRA | Resource Conservation and Recovery Act |
| ROI | radius of influence |
| SAR | sodium adsorption ratio |
| SCAQMD | South Coast Air Quality Management District |
| SVE | soil vapor extraction |
| SVOC | semi-volatile organic compound |
| TCA | trichloroethane |
| TCE | trichloroethylene |
| TMB | trimethylbenzene |
| TOC | total organic carbon |
| TPH | total petroleum hydrocarbon |
| UFA | Unsaturated Flow Analysis |
| U.S. EPA | United States Environmental Protection Agency |
| USGS | U.S. Geological Survey |
| UVB | Unterdruck-Verdampfer-Brunnen (vacuum vaporizer well) |
| VC | vinyl chloride |
| VOA | volatile organic analysis |
| VOC | volatile organic compound |

# VOLUME 1

# TECHNOLOGY SCREENING

# 1. Introduction

Groundwater circulating wells (GCWs) are a developing environmental remediation technology originally developed for removal of hydrocarbon contamination from groundwater. Recent advances in the technology have expanded the application to a wide range of contaminants, and new system configurations often incorporate simultaneous vadose zone treatment. Although GCWs have been successfully applied at many sites, the technology continues to develop with improvements in design and operation efficiency.

## 1.1 Protocol Objective

The objective of this protocol is to assist the user in all aspects of implementing the GCW technology for remediation at any given site. The protocol was written to guide the user through an effective and systematic process to accomplish the following:

- evaluate the feasibility of using a GCW system at a contaminated site
- select the appropriate system configuration based on the contamination and hydrogeologic properties at the site
- develop a site-specific design
- install, operate, and monitor the system
- evaluate the performance of the system based on operational data.

The protocol is arranged in the following three sections.

Section I: Technology Screening
Section II: Technology Guidance Implementation
Appendix: Current Vendor Information and Case History Information

Section I provides the user with an introduction to the GCW technology and describes a screening process that can be used to determine if a site is a candidate for this technology. Section II contains a guide summarizing GCW design, modeling, monitoring, and system evaluation procedures. The Appendix contains vendor-provided information describing some of their successful GCW installations. This information is included to provide potential GCW users with a reference to GCW applications for various contaminant and/or geologic settings, not to promote any specific vendor or vendor system.

The protocol is not a design document but rather a guide to help the user understand the GCW technology. This document can be used to ensure that consistent procedures are followed for effective GCW implementation and to ensure the achievement of remediation goals.

## 1.2 Background

Historically, aquifer restoration has focused on the use of pump-and-treat technologies that entail installing a network of pumping wells within and around a contaminant plume, and pumping water to the surface for treatment. The pumped water serves to transport contamination to the well for subsequent removal from the aquifer. The rate of remediation is limited by mass transfer and sorption/desorption kinetics. Some pump-and-treat systems have been designed with a reinjection capability for facilitating groundwater movement and/or for introduction of amendments such as nutrients, electron donors or acceptors, or microorganisms. The addition of these amendments has resulted in varying levels of success.

Pump-and-treat technologies have some general limitations. They can require pumping large quantities of groundwater from an aquifer. If contaminant concentrations in the pumped water exceed regulatory levels, the water requires treatment prior to discharge. Common treatment technologies coupled to pump-and-treat systems include aboveground air stripping, activated carbon adsorption, and biological treatment. Treated water can be discharged either to a sanitary or industrial sewer line, depending on local permitting requirements. Reinjection to the aquifer may be possible if the treated water meets local regulatory standards and an injection permit is obtained. These permits are difficult to obtain for reinjection into drinking-water source aquifers.

Major costs of pump-and-treat systems are associated with lifting the water and with the aboveground treatment processes required to achieve stringent treatment levels. At sites with deep groundwater, where large pumps with greater lifting capacities are required, the energy costs associated with lifting the water to the treatment unit can be a significant portion of the remediation cost. On the other hand, the costs associated with aboveground treatment may predominate when large quantities of water are pumped and/or a high degree of treatment is required.

Alternative methods to conventional pumping and treating are being developed that provide in situ treatment, thus eliminating the need for groundwater withdrawal and aboveground treatment. Air sparging, enhanced in situ aerobic and anaerobic biodegradation, chemical and biological barrier technologies, and GCWs are examples of such alternatives. This protocol focuses on the use of GCWs for aquifer restoration and site remediation.

Groundwater circulating well systems (GCWs) are designed to create a 3-dimensional circulation pattern in an aquifer by drawing groundwater to the well, pumping the water through the well, then reintroducing the water into the aquifer without pumping it above ground. Distinct circulation patterns are established depending on both the operational mode of the GCW and the hydrogeologic conditions of the site. GCWs can be configured with upward in-well flow (upflow) or downward in-well flow (downflow) depending on site requirements. Figure 1 is a generalized schematic showing the flow schemes for each type.

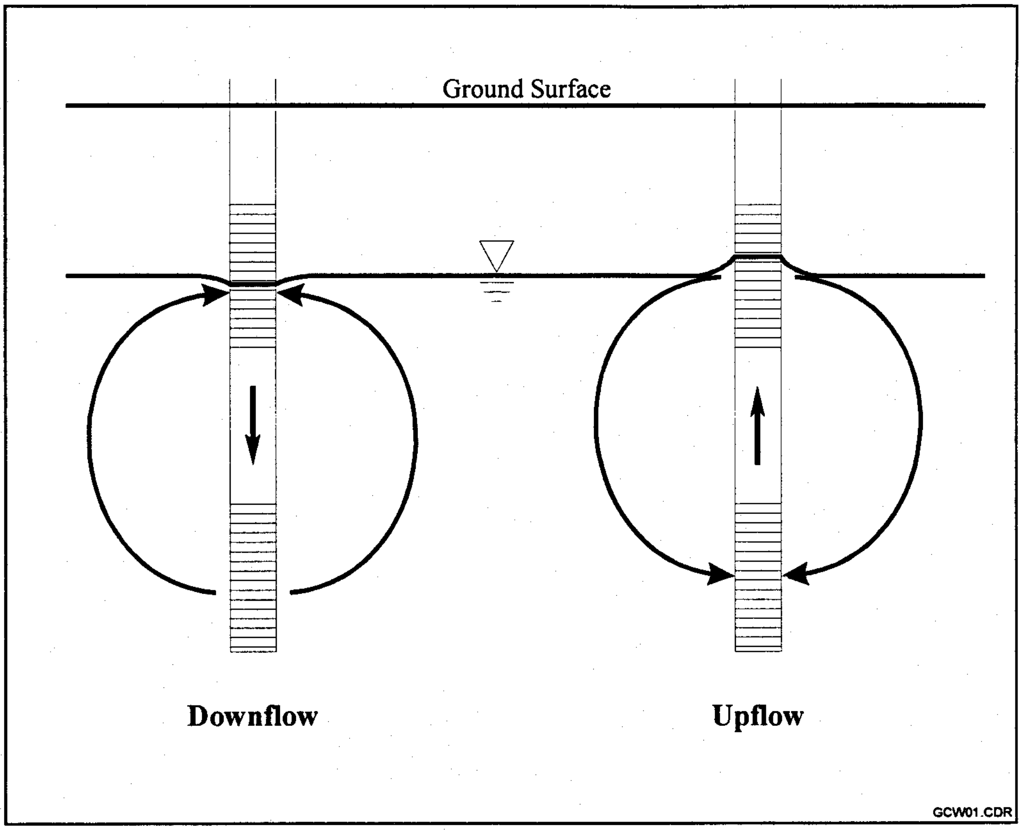



**Figure 1. Generalized Schematic of Upflow and Downflow Groundwater Circulating Wells.**

Figure 2 illustrates ideal circulation patterns that would be established with an upflow GCW system as a function of horizontal groundwater flow velocities (Herrling et. al., 1991). Figure 2a shows that under ideal conditions, and in the absence of background groundwater flow, the circulation pattern forms a symmetrical ellipsoid around the well. Figures 2b and c show that as the horizontal component of the background groundwater velocity increases, the streamlines are skewed and the symmetry is lost. By comparing Figures 2b and c, it can be seen that increasing background flow velocities causes a decrease in the radius of the flow field. This has a direct impact on the GCW-induced circulation cell and dictates the GCW spacing required for effective system design. Circulation patterns for downflow GCW systems would be mirror images in the x-plane of the patterns shown in Figure 2.

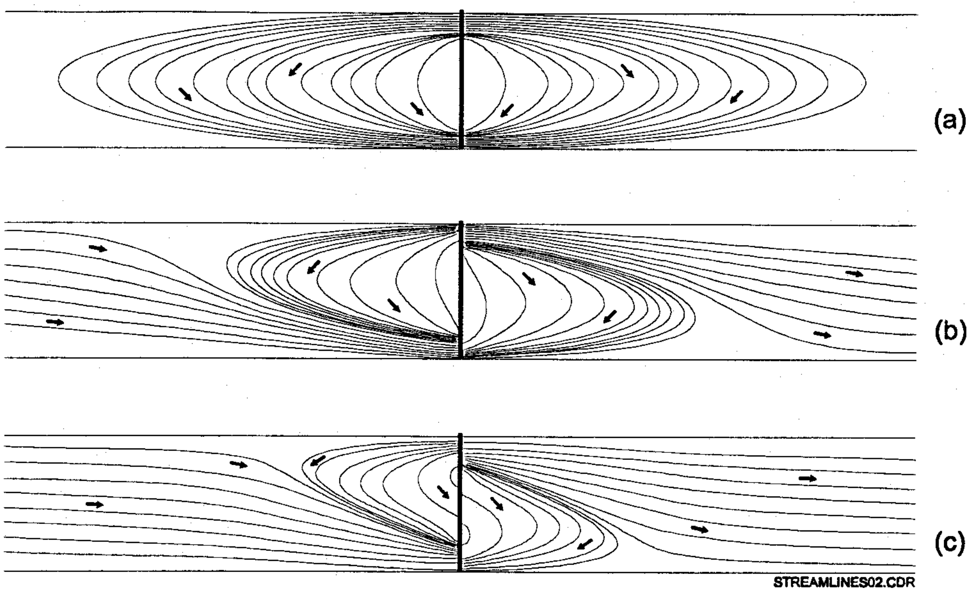



**Figure 2. Idealized Circulation Pattern Around an Upflow GCW System with Horizontal Groundwater Velocities of (a) 0.0 m/day, (b) 0.3 m/day, and (c) 1.0 m/day (reprinted from Herrling et al., 1991).**

Figure 3 illustrates the results of numerical simulations showing the effect on the circulation pattern with two GCW systems placed close enough to affect each other significantly (Herrling et al., 1994). This type of well spacing may be required to treat a targeted volume effectively or to develop an effective capture zone. The strong vertical circulation between the two wells may prove beneficial for highly contaminated areas near a source (Herrling et. al., 1994).

GCW systems are designed to provide treatment inside the well, in the aquifer, or a combination of both. For effective in-well treatment, the contaminants must be adequately soluble and mobile for transport by circulating groundwater. Current methods for in-well treatment include air stripping, activated carbon adsorption, and biodegradation. The selection of which in-well method will be used is contaminant-, site-, and vendor-specific. Contaminants that cannot be mobilized effectively to the well need to be treated in the aquifer. Most commonly, in situ treatment is achieved through biodegradation. Currently, GCW systems promote aerobic biological activity; however, anaerobic systems are under development. Often, there is a combination of both in-well and in-aquifer treatment, with the relative percentages of removal by each mechanism being contaminant-, site-, and GCW configuration-specific.

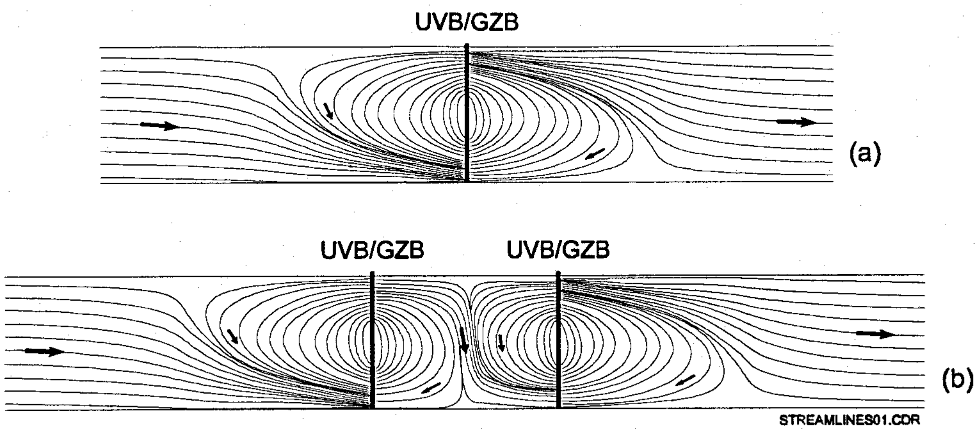



**Figure 3. Numerical Simulation Showing the Circulation Pattern around Two GCW Systems (Herrling et al., 1994).**

Many GCW configurations incorporate air lift pumping to facilitate groundwater circulation, and/or air stripping to remove contaminant from the groundwater passing through the well. These systems transfer volatile contaminants from the aqueous phase to the vapor phase. The off-gas containing contaminant vapors can often require some level of treatment prior to discharge. Treatment can be achieved in situ by direct injection of the off-gas into the vadose zone. More often, the off-gas is treated above ground with any of a variety of processes.

Depending on the specific application, GCW systems may offer several advantages over conventional pump-and-treat technologies. One advantage is that treatment of the contaminated groundwater takes place below grade and does not require pumping groundwater out of the ground. Eliminating the need to pump groundwater to the surface is an attractive feature of GCWs for two main reasons. First, treating groundwater in situ may eliminate the discharge permitting requirements associated with conventional pump-and-treat systems. Second, by achieving treatment below ground, the need to lift groundwater can be reduced or eliminated and significant energy cost savings may be realized, especially at sites with deep water tables.

Another advantage of GCW systems over conventional pumping and treating is that they induce a groundwater circulation zone that "sweeps" the aquifer. Pump-and-treat systems cause drawdown around the well, leaving contaminated zones that are not treated. In addition, pump-and-treat systems inherently draw water along preferential pathways in the subsurface, leaving contamination in lower permeability units. Because of this, pump-and-treat systems often become diffusion limited. GCW systems do not cause drawdown and they promote a circulation with both horizontal and vertical flow components that can cut across lower permeable units.

Another potential benefit that can be realized with a number of GCW designs is the simultaneous remediation of contamination above and below the water table. This is achieved by coupling the saturated zone treatment with soil vapor extraction (SVE) or bioventing. Coupling to SVE results in the contaminants being entrained in the vapor flow to the GCW where they are removed along with the GCW system off-gas for aboveground treatment. Incorporating bioventing into the design can provide for in situ destruction through biodegradation of both residual soil and GCW off-gas contamination.

These are attractive features that suggest consideration should be given to GCWs during the technology selection process for remediation of contaminated groundwater. However, the applicability of the GCW technology is site specific and engineering decisions must be made based on site-specific criteria before selecting a GCW system for remediation at any site.

**1.3 Overview of the Groundwater Circulation Well Technology**
There are a number of different configurations of GCWs available for a variety of applications; however, the basic operating mechanisms of all configurations are similar. All GCWs function by moving water through a well or borehole placed within a contaminated zone within an aquifer. Contaminated water enters the GCW through the influent section. The water is treated and/or amended within the GCW, then reinjected into the aquifer through the effluent section of the well. All GCWs circulate the water in situ without pumping it above ground. The specifics of the well design, method of pumping, and method of treatment vary by configuration and are selected based upon site- and application-specific requirements. These following sections introduce the basic principles of operation of GCWs.

**1.3.1 Methods for Moving Groundwater.** GCW designs are available that move water in either an upflow or downflow direction. The direction depends on the water pumping mechanism which is often configuration specific. The desired mode of operation depends on the contaminant type and distribution, and on the other hydrogeologic factors discussed in Section II-2.2. The required mode of operation will dictate the selection of the specific GCW configuration.

**1.3.1.1 Air Lift Pumping.** One of the most common methods GCWs use to move groundwater is air lift pumping. Air lift GCWs have an air line that extend to some depth below the water level in the well system. As air is injected, it mixes with the water and causes a decrease in the specific gravity of the fluid (Powers, 1992). The difference between the weight of the air-water fluid within the well and the water in the formation outside the well causes the water in the well to rise. As the water rises, the displaced water in the bottom of the well is replaced by water drawn in from the formation. More detailed information on air lift pumping can be found in Stepanoff 1965, Gaver and Aziz 1972, Perry et al. 1984, Morrison et al. 1987, Gvertzman and Gorelick 1992, and/or Francois et al. 1996.

Depending on the specific GCW configuration, the pumped water is reintroduced into the aquifer either directly through the upper well screen or through a subsurface infiltration gallery. Air lift GCW systems, usually operate in an upflow mode. A downflow airlift system has been proposed, but it has not been demonstrated effective in any field application. A more detailed discussion of the individual GCW configurations currently available is provided in the Appendix.

**1.3.1.2 Mechanical Pumping.** Another method used to facilitate groundwater movement is mechanical pumping. Installing mechanical pumps into GCW units allows the systems to be operated in either an upflow of downflow mode. Mechanical pumps are incorporated into dual screen well designs that include a packer to separate the two screened sections. Packing off the screens allows for water movement from one screened section to the other without short circuiting within the well. Descriptions of several GCW systems that utilize mechanical pumping can be found in the Appendix.

**1.3.2 Common methods for treating groundwater.** One of the attractive features of GCW systems is that groundwater treatment occurs in situ, within the well volume and within the aquifer, thus eliminating the need for pumping groundwater above ground. For in-well treatment, various treatment technologies including air stripping, carbon adsorption, and biological degradation have been incorporated into GCW designs. In the aquifer outside the well, contaminants are removed through desorption and transport to the well, and through biodegradation supported through the delivery of nutrients and/or electron acceptors/donors in the circulated water. Brief descriptions of these processes are presented in the following sections.

**1.3.2.1 In-Well Processes.** The most common method for in-well groundwater treatment within GCW systems is air stripping. Often, this process is coupled with the air lift pumping mechanism described in Section 1.3.1.1. Other in-well groundwater treatment processes include activated carbon adsorption, biological treatment, or air stripping coupled with mechanical pumping.

**1.3.2.1.1 Air Stripping.** Air stripping is the most commonly applied process for achieving in-well groundwater treatment in GCWs. This process serves both to remove volatile organic compounds from the water, and to aerate the water prior to discharge from the well system. Air stripping is a proven, effective technology for aboveground treatment of water and wastewater containing volatile contaminants. In GCW systems, the air stripping process is often facilitated by the air injected to drive the air lift pumping which in turn drives the groundwater circulation. For systems that couple air lift with air stripping, it is necessary to balance the operation of these two mechanisms for optimum performance.

Air stripping is a phase transfer process during which volatile contaminants are exchanged from the aqueous phase to the gaseous phase. The partitioning between the phases is a function of the temperature of the two phases, the total pressure in the system, and the molecular interactions occurring between the contaminant and the water (Montgomery, 1985). Henry's law describes the partitioning of the contaminant between the water and gas phases at equilibrium. Combining Henry's law with Dalton's law results in Equation 1 (Montgomery, 1985):

$$Y_i = \frac{X_i H_i}{P_T} \qquad (1)$$

where: $Y_i$ = the mole fraction of the gas phase
$X_i$ = the mole fraction of the contaminant in the water phase
$H_i$ = the Henry's law constant
$P_T$ = the total system pressure

If both the Henry's constant of a contaminant and the total pressure are known, the above relationship can be used to determine the equilibrium partitioning between gas and water phases during air stripping. In general, compounds with a higher Henry's constant are more easily stripped from water than compounds with lower Henry's constants. Table 1 contains Henry's constants for a list of organic compounds that can be readily air stripped (Montgomery and Welkom, 1990).

**Table 1. Henry's Constants (atm·m$^3$/mol) for Selected Organic Compounds**

| Compound | Henry's Constant | Compound | Henry's Constant | Compound | Henry's Constant |
|---|---|---|---|---|---|
| Benzene | 0.00548 | Chloroform | 0.0029 | perchloroethylene | 0.0153 |
| Toluene | 0.00674 | carbon tetrachloride | 0.0302 | trichloroethylene | 0.0091 |
| Ethylbenzene | 0.00868 | Chlorobenzene | 0.00445 | 1,1-dichloroethylene | 0.021 |
| o-xylene | 0.00535 | 1,2-dichlorobenzene | 0.0024 | cis-1,2-dichlrorethylene | |
| m-xylene | 0.0063 | 1,3-dichlorobenzene | 0.0047 | trans-1,2-dichloroethylene | 0.00674 |
| p-xylene | 0.0063 | 1,4-dichlorobenzene | 0.00445 | vinyl chloride | 0.056 |
| Naphthalene | $4.6 \times 10^{-4}$ | | | | |

Effective operation of GCW systems that utilize air lift pumping and in-well air stripping of VOCs requires a balance between the pumping and stripping efficiencies. Typically, the optimum air injection rates for air lift pumping and air stripping do not coincide. More detailed information on the theory of air stripping and design procedures for air stripping systems can be found in Canter and Knox 1986, Montgomery and Wellon 1990, and Gvirtzman and Gorelick 1992.

**1.3.2.1.2 Activated Carbon Adsorption.** GCW configurations are available that utilize activated carbon adsorption as the in-well treatment process. Activated carbon is commonly used in water and wastewater treatment, usually as a polishing step, and also is used for vapor-phase treatment of off-gas from unit processes such as air strippers, soil vapor extraction, and bioslurping systems. In GCWs, granular activated carbon (GAC) canisters are placed within the well. As the contaminated groundwater is pumped through the GCW, it passes through the GAC where the contaminants are adsorbed to the carbon. The clean water is discharged through the effluent portion of the GCW.

As with air stripping, activated carbon adsorption is a phase transfer process with the contaminant being transferred from the aqueous phase to the solid phase. Adsorption is the physical/chemical process of accumulating contaminants at a solid/liquid interface. The contaminant is referred to as the adsorbate and the activated carbon as the adsorbent.

Adsorption occurs when there are forces that attract the adsorbate to the surface of the adsorbent. These forces can result in two types of adsorption, termed physical or chemical adsorption. Physical adsorption is promoted by weaker electrostatic forces such as hydrogen bonding and London-van der Waals forces and by hydrophobic interactions. Oftentimes, physical adsorption is a reversible process and a "feedback" phenomenon can occur if the adsorbate concentration in the aqueous phase decreases to below the equilibrium concentration.

Chemical adsorption, referred to as "chemisorption," is promoted by stronger electrostatic forces that more resemble covalent or electrostatic bonding between two atoms (Montgomery, 1985). Chemisorption typically occurs at specific sites on the surface of the activated carbon, and may be strongly influenced by specific functional group types and densities. In general, chemisorption is less reversible than physical sorption.

The adsorptive capacity of an activated carbon is dependent on the properties and concentration of the adsorbate and on temperature and pH (Metcalf and Eddy, 1991). Table 2 lists compounds that adsorb readily onto activated carbon (Montgomery, 1985). For more in depth information on activated carbon adsorption theory and design, see Montgomery 1985, and/or Metcalf and Eddy 1991.

**Table 2. Compounds Readily Adsorbed onto Activated Carbon.**

| Class | Example Compounds |
|---|---|
| Aromatic Solvents | Benzene |
| | Toluene |
| | Ethylbenzene |
| | Xylenes |
| | Nitrobenzenes |
| Chlorinated Aromatic Compounds | polychlorinated biphenyls (PCBs) |
| | Chlorobenzenes |
| | chlorinated phenols |
| Chlorinated Aliphatic Compounds | carbon tetrachloride |
| | Trichloroethylene |
| High Molecular Weight Hydrocarbons | Gasoline |
| | Diesel |
| | JP-4, JP-5, and JP-8 |

**1.3.2.1.3 Biological Treatment.** GCWs are available that achieve in-well treatment of contaminated groundwater through biodegradation by incorporating a bioreactor in the GCW design (see Appendix). The reactor utilizes biofilm technology, and is available in either a spiral wound membrane of an activated carbon support medium configuration. The indigenous microorganisms within the aquifer are allowed to colonize the surface of the activated carbon as groundwater is circulated through the well. Once a sufficient population of these microorganisms develops, biodegradation can occur as the water passes through the reactor. This type of GCW has been used to successfully treat groundwater contaminated with PAHs.

**1.3.2.2 In Situ Processes.**

**1.3.2.2.1 Desorption and Transport.** One of the overall objectives of GCW systems is to transport the contamination from the formation to the well for treatment. The rate at which a contaminant can be transported to the well is dependent on its solubility and on the physical and chemical characteristics of the formation(s) through which the solubilized contaminant migrates on its way to the well. Typically, the more soluble a compound, the faster it can be transported to the well. Contaminant migration can be slowed in tight formations or when organic matter is present.

**1.3.2.2.2 Biological Degradation.** Another objective of GCW systems is to remediate contamination in the formation without transporting it to the well. The primary mechanism for achieving in-place treatment is biological degradation. Typically, water entering the GCW is anoxic. As the water is air lift pumped and aerated, the oxygen can achieve saturation levels (approximately 8 mg/L). The oxygenated water is circulated back into the formation where the oxygen can support aerobic biodegradation of contaminants. Unfortunately, delivering 8 mg of oxygen per liter of water means that a large volume of water must be pumped to provide the oxygen required to support biodegradation of a given mass of contaminant. Recent evidence suggests that although the oxygen delivery capacity of some GCW configurations may be limited, the induced water circulation may be effective at enhancing anaerobic biodegradation. This phenomenon has not been documented.

**1.3.2.2.3 In Situ Oxidation.** One available GCW configuration combines ozone and air injection with a downflow system (see Appendix). The ozone and entrained air are forced into the formation where the ozone can attack the contaminant. The system includes an air sparging point below the well casing that aerates the water below the well and transports contaminants up into the ozonated region of the formation for treatment.

**1.3.3 Common methods for treating system off-gas.** GCW systems that utilize air stripping as the mechanism for removing contaminants from groundwater produce vapor (off-gas) containing the transferred contaminants. The vapors can be biologically treated in situ through direct injection into the vadose zone, or ex situ by a number of treatment processes. The selection of vapor treatment is dependent on both contaminant type and contaminant concentration. The following sections contain descriptions of vapor treatment alternatives that can be used with GCW systems.

**1.3.3.1 In Situ Biodegradation.** In situ bioremediation of the GCW vapor emissions through direct injection of the off-gas into the vadose zone can be a cost-effective and environmentally sound treatment option. In effect, the GCW mimics bioventing with the GCW serving as the vent well. This coupling can serve to remediate residual vadose zone contamination as well as the contaminant in the introduced vapor. Direct injection of off-gas can offer the advantages of low surface emissions and no point-source generation.

Direct injection is accomplished by designing the GCW system so that the vapor containing stripped contaminants passes from the well directly into the vadose zone soil without being pulled above-grade. The vapor must contain sufficient residual oxygen to support aerobic biological activity to facilitate in situ biodegradation of the contaminants. In situ respiration and soil gas permeability data must be available for the site. These data indicate the expected biodegradation rate and radius of influence (ROI) that are required to determine the design capacity for the direct injection GCW system. The soil volume available must be sufficient to accept the off-gas airflow and allow biodegradation of the contaminant mass flow in the off-gas.

Direct injection GCW systems must be designed and located to ensure that the injection process is destroying the contaminants rather than increasing contaminant migration. After injection is established, surface emission testing can be performed to ensure that contaminants are not escaping at the site surface. Soil gas monitoring must be performed to ensure that contaminant migration is not being increased. Monitoring of migration is particularly important at sites where air extraction may be necessary to prevent migration into nearby buildings.

**1.3.3.2 Ex Situ Processes.** In instances where in situ processes are not feasible, several treatment methods are available for the aboveground treatment of contaminated off-gas. These methods include adsorption, catalytic oxidation, thermal oxidation, and biofiltration. Before the appropriate ex situ process can be selected permitting requirements and regulations governing discharge limits must be investigated. The following sections provide an overview of the regulation and commercially available applications of each of these technologies.

**1.3.3.2.1 Regulation.** The U.S. EPA reported that the results of a state-by-state telephone survey conducted in July and August, 1989 revealed that nearly half (24 of 51) of the states had no statewide air discharge standards for SVE systems. These states relied on federal standards for such discharges (U.S. EPA, 1991, EPA/540/2-91/003). Many of the general emission-source laws were written primarily for large sources such as power plants. GCW systems (which are small sources by comparison) may not require off-gas treatment if only small quantities of VOCs are emitted. The 1989 survey illustrated how widely states vary in their air emission regulations, from little or no formal regulation to detailed regulation of specific chemical contaminant releases.

In 1990, the Clean Air Act was amended, resulting in tighter controls of air discharges; however, the amendment did not mandate that states change their permitting requirements for off-gas from remediation systems. Some states continue to base their standards on the concentration at the nearest receptor, whereas others treat each site on a case-by-case basis. While the details of VOC discharge regulation have changed since 1989, the general trends of local control and widely varying regulatory approaches and discharge limits are still observed. Permitting requirements are site specific and it is necessary to apply for discharge permits on a case-by-case basis.

**1.3.3.2.2 Adsorption.** Adsorption refers to the process by which molecules collect on and adhere to the surface of an adsorbent solid (U.S. EPA, 1988, EPA/530/UST-88/001). This adsorption is due to chemical and/or physical forces. Surface area is the critical factor in the adsorption process, because the adsorption capacity is proportional to the surface area. Commercially available adsorbents include activated carbon and synthetic resins.

Granular activated carbon is the most commonly used vapor-phase treatment method. Activated carbon adsorbents provide a high surface area in a low-unit-cost material due to the carbon's complex internal pore structure. Commercially available GAC typically has a surface area of 1,000 to 1,400 $m^2$/gram. As a rule of thumb, the adsorptive capacity of activated carbon for most hydrocarbons in the vapor stream is about 1 lb hydrocarbon:10 lb activated carbon. The cost of activated carbon is about $3/lb (all costs included, not just carbon purchase), so the cost of activated carbon treatment can be estimated roughly as being about $30/lb of hydrocarbon to be treated. Estimated costs for vapor treatment with GAC are given in Table 3.

**Table 3. Estimated Costs for Off-Gas Treatment with Granular Activated Carbon**

| Cost Item | Cost with Inlet Concentration of 1 ppmv | Cost with Inlet Concentration of 3,500 ppmv | Cost with Inlet Concentration of 14,000 ppmv |
|---|---|---|---|
| Capital cost ($) | 0.00 | 0.00 | Not applicable |
| Monthly operating cost ($/month) | $32.30 | $113,000.00 | Not applicable |
| Total cost for 6-month operation ($) | $193.80 | $678,000.00 | Not applicable |

Note: Figures are based on an off-gas flow of 65 scfm.

GAC is a cost-effective organic vapor treatment method for a wide range of applications due to its relative ease of implementation and operation, its established performance history in commercial applications, and its applicability to a wide range of contaminants at a wide range of flow rates. Many vendors sell or lease prefabricated, skid-mounted units that can be put into operation with a few days' notice. However, carbon adsorption is economical only for lower mass removal rates. When the vapor concentration is high, carbon replacement requirements may be prohibitively expensive.

The adsorption capacity of the carbon depends on several factors, including influent vapor temperature, relative humidity, and most importantly, the influent VOC types and concentrations. Isotherms, which show the mass of contaminants that can be adsorbed per unit mass of carbon, are available to predict the contaminant-specific adsorption capacity for a specific type of carbon. GAC generally has a high affinity for volatile organic molecules, such as hydrocarbons or chlorinated compounds. These volatile organics are the types of compounds typically removed by airflow through organic-contaminated soil; however, some hydrocarbons such as isopentane are adsorbed poorly.

Although GAC has a very high surface area for adsorption of contaminants, the mass of contaminants removed may exceed the carbon's capacity. At sites requiring high organic-contaminant mass removal rates due to high concentration, high flowrate, or both, the adsorption capacity of the carbon may be quickly exhausted. Replacement and disposal of spent carbon can become expensive. The cost of disposal of spent carbon will be particularly high if hazardous solvents sorbed on the spent carbon result in the entire spent carbon volume being identified as a Resource Conservation and Recovery Act (RCRA)-listed or RCRA-characteristic waste.

High relative humidity in the incoming vapor stream limits the effectiveness of and increases the cost of vapor treatment with GAC. Water vapor preferentially occupies adsorption sites, thereby decreasing the capacity of the carbon to remove contaminants from the air stream. If necessary, entrained water must be removed from the incoming vapor stream by an air/water separator. Vendors typically recommend that the relative humidity of the off-gas stream should be below 50% prior to entering the GAC adsorber. Reduction of relative humidity usually is achieved by pre-warming the air. However, preheating to reduce the effect of water vapor also reduces the effective capacity of the carbon, so a trade-off is involved in selecting the preheat temperature.

Specialized resin adsorbents have been developed and are now entering commercial application for treatment of organic vapors in off-gas streams. These synthetic resin adsorbents have a high tolerance to water vapor. Air streams with a relative humidity greater than 90% can be processed with little reduction in the adsorption efficiency for organic contaminants. The resin-adsorbents are amenable to regeneration on site. Skid-mounted modules are available consisting of two resin adsorbent beds. The design allows one bed to be on-line treating off-gas while the other bed is being regenerated. During desorption, all of the organic contaminants trapped on the resin are removed, condensed, and transferred to a storage tank.

**1.3.3.2.3 Biofiltration.** Vapor-phase bioreactors are an effective method for treating a variety of gas-phase organic contaminants and have been successfully employed to treat off-gas from remediation processes including soil vapor extraction, bioslurping and GCW systems (Connolly et al., 1995). The effective treatment of influent vapor concentrations ranging from 50 to 5,000 ppmv by bioreactors has been reported in the literature (U.S. EPA, 1994, EPA/542-R-94-003). Because GCW system off-gas contains high percentages of oxygen, vapor phase bioreactors are more suited for contaminants that are readily biodegraded under aerobic conditions. SBP Technologies, Inc. is participating in a EPA SITE Program demonstration of a GCW system with a vapor-phase bioreactor off-gas treatment system at a site in Sweden, New York. The site is contaminated with trichloroethlyene (TCE), acetone, perchloroethylene (PCE), dichloroethylene (DCE), and toluene. Because PCE is not a candidate for aerobic degradation and the off-gas from the reactor may contain levels of non-degraded contaminants, the vapor-phase bioreactor is followed by activated carbon treatment to remove any residual compounds prior to air discharge. Because biofiltration is an innovative technology, insufficient data precluded a detailed cost or performance comparison with the more conventional off-gas treatment technologies.

**1.3.3.2.4 Thermal Oxidation.** Thermal oxidation units use high temperatures to drive the oxidation of organic contaminants in an off-gas stream. Three commercially available thermal oxidation designs include open flame, flameless, and internal combustion engine systems. Open flame oxidation converts hydrocarbon compounds to carbon dioxide and water by direct thermal oxidation. Complete destruction of contaminants requires high temperatures, typically 1,200 to 1,600°F, and/or long residence times. The flameless oxidation process converts hydrocarbon compounds to carbon dioxide and water by passing an off-gas stream through a heated ceramic matrix. The matrix geometry and uniform high temperature of the matrix are reported to give good destruction efficiency for organic vapors in air, without using an open flame. Internal combustion engine treatment accomplishes destruction of organic contaminants by oxidation in an ordinary industrial or automotive engine with its carburetor modified to accept vapors rather than liquid fuel. A catalytic converter is an integral component of the system, providing an important polishing step to reach the low discharge levels required by many regulatory agencies. Thermal oxidation becomes cost competitive when the inlet vapor concentration approaches 25% of the LEL. For more information on thermal oxidation processes, system design and costs, see Battelle, 1997.

**1.3.3.2.5 Catalytic Oxidation.** Catalytic oxidation is a thermal treatment process that uses a catalyst to increase the oxidation rate of organic contaminants in an off-gas stream, allowing acceptable destruction efficiency at a lower temperature than thermal oxidation. In catalytic oxidation, the off-gas is heated and passed through a combustion unit where the gas stream contacts the catalyst. Without undergoing a chemical change itself, the catalyst increases the oxidation reaction rate by adsorbing the contaminant molecules on the catalyst surface. Sorption phenomena serve to increase the local concentration of organic contaminants at the catalyst surface and, for some organic contaminants, reduce the activation energy of the oxidation reaction. Increased concentration and reduced activation energy increase the rate of oxidation of the organics (Kiang, 1988).

Treating off-gas containing halogenated compounds, sulfur-containing compounds, or nitrogen-containing compounds will deactivate a conventional catalyst. Deactivation results from chemical interactions between the catalyst metal with halogens or strong sorption of $SO_X$ and $NO_X$ onto the catalyst. Catalysts specially designed for treatment of chlorinated compounds by catalytic oxidation are available on the market but are more expensive than catalysts for treating petroleum hydrocarbons. The oxidation unit typically will require scrubbing to remove acid gases formed when treating halogenated compounds (Buck and Hauck, 1992).

## 1.4 Factors that Affect the Applicability of the Groundwater Circulation Well Technology

### 1.4.1 Nature of Contaminant

Important factors affecting the feasibility and selection of GCW processes include the type contaminant being remediated, its chemical characteristics, physical distribution in the environment, and ability to be treated chemically or biologically. Specifically, the mass transfer and destruction mechanisms of the specific GCW configuration must be capable of moving the contaminant to the well for treatment or removal and/or of supporting biodegradation of the contaminant in situ. It should be noted that for less mobile contaminants, in situ biodegradation could account for the majority of treatment within the zone of influence of GCWs. The majority of removal for more mobile contaminants is accomplished by moving the contaminant from various reservoirs in the zone of influence to the treatment unit within the well. Contaminant characteristics will impact the effectiveness of both the transport of the contaminant to the GCW, and the removal efficiency of the treatment unit inside the GCW.

**1.4.1.1 Contaminant type(s).** The class of chemical to which the contaminant compound belongs is an important factor affecting the remediation technology selection process. Classes of compounds commonly share important characteristics; that is, they often have similar solubilities, vapor pressures, and reactivities. Table 4 lists several classes of contaminants and some characteristic properties of each class.

**Table 4. Contaminant Types and Properties**

| Contaminant Type | Example | Common Property |
|---|---|---|
| Heavy Metals | Lead | Toxicity |
| Chlorinated Solvents | TCE | DNAPL |
| Radionuclides | Radium | Radioactive |
| PAHs | Naphthalene | Lower Solubility |
| Petroleum Hydrocarbons | JP-4 | Biodegradable LNAPLS |

Remediation systems are often designed to remove a class of chemical compounds. For example, a particular GCW design may be targeted to remove heavy molecular weight petroleum hydrocarbons, and would therefore be a candidate technology for the removal of diesel fuel, heavy jet fuel (JP-5), and heavier utility fuels like bunker fuel. The GCW design that is appropriate for removing heavy hydrocarbons may not be appropriate for removing a contaminant that belongs to another class of compounds, because GCW designs often exploit physical/chemical properties common to a specific classification to effect their removal. It is important to review contaminants' physical/chemical properties prior to selecting and designing a GCW system so that opportunities for contaminant property exploitation can be identified.

To design a remediation system for contaminant removal, it is essential to consider the physical and chemical properties, and the biodegradability of the contaminant(s). For example, a contaminant that is not easily volatilized from the aqueous phase should not be considered for removal with a technology that includes air stripping as a primary treatment process. If the contaminant is soluble and can be transported to the well, in-well carbon absorption or biodegradation may be more appropriate. Table 5 lists several contaminant properties that must be considered during the GCW technology selection and system design process.

**Table 5. Contaminant Properties to be Considered for Remediation Technology Selection and System Design**

| Property | Units | Impact |
|---|---|---|

| | | |
|---|---|---|
| Solubility | g/L | Aqueous mass transfer and transport to the well |
| Volatility, (vapor pressure) | atm | Gaseous mass transfer and effectiveness of air stripping |
| Octanol/Water Partitioning Coefficient ($K_{OW}$) | ML water/mL octanol | Affinity for organic material transport to the well |
| Henry's Constant | atm $m^3$/mol | Vapor-phase concentration air stripping efficiency |
| Molecular Weight | g/mol | Stoichiometry |
| Density | g/$cm^3$ | Environmental distribution |
| Biodegradability | y/n aerobic/anaerobic | Technology selection |
| Bioaccumulated | y/n | Risk assessment |
| Toxicity | LD50 or PEL | Health and safety |

The solubility, $K_{OW}$, Henry's constant, and/or the biodegradability of the contaminant will influence the treatment process selection for contaminant removal. More volatile compounds, such as BTEX, can be easily air stripped from the circulated groundwater. Compounds such as PAHs, phenolic compounds, or heavy metals are less volatile and air stripping would not be the best process for removal of these classes of contaminants.

A contaminant with a relatively high $K_{OW}$ value will have difficulty being mobilized in the groundwater and transported to the GCW. This is especially true in aquifers with higher organic content, with these contaminants the majority of treatment may require in situ biodegradation treatment. Whether the compound can be biodegraded aerobically or anaerobically will determine whether air-saturated water or organic nutrients should be added to the circulation water, respectively. BTEX compounds, for example, are easily aerobically biodegraded while PCE can only be anaerobically biodegraded. TCE can be aerobically biodegraded, but only cometabolically, which requires the addition of a cosubstrate such as methane, phenol, or toluene. The metabolic pathways to be exploited for contaminant biodegradation must be well understood before selecting and/or designing a GCW system. Biological degradation processes for some common halogenated and nonhalogenated contaminants are shown in Table 6, as is a list of the types of metabolism for some important environmental contaminants.

**1.4.1.2 Contaminant Phase/Distribution.** Contaminants released into the environment may be present in any or all of four phases in the geologic media (Battelle, 1995):

- sorbed to the soils in the vadose or saturated zones
- in the vapor phase in the vadose zone
- in free-phase form either floating on the water table as LNAPL, as residual saturation in the vadose zone, or submerged within or at the bottom of the aquifer as DNAPL
- in the aqueous phase dissolved in pore water in the vadose zone or dissolved in the groundwater.

**Table 6. Metabolic Pathways for Degradation of Environmental Contaminants**

| Compound | Aerobic Growth Substrate | Aerobic Cometabolism | Anaerobic Dechlorination |
|---|---|---|---|
| PCE | | | ✓ |
| TCE | | ✓ | ✓ |
| *c*-DCE | | ✓ | ✓ |
| 1,1-DCE | | ✓ | ✓ |
| Vinyl Chloride (VC) | ✓ | ✓ | ✓ |
| 1,1,1-Trichloroethane (TCA) | | ✓ | ✓ (a) |
| 1,1,2-TCA | | ✓ | ✓ (a) |
| 1,1- Dichloroethane (DCA) | ✓ | ✓ | ✓ (a) |
| 1,2-Dichlorobenzene (DCB) | ✓ | | ✓ |
| 1,3-DCB | ✓ | | ✓ |
| 1,4-DCB | ✓ | | ✓ |
| Chlorobenzene | ✓ | | |
| 1,2,4-Trimethylbenzene (TMB) | ✓ | | |
| 1,3,5-TMB | ✓ | | |
| Methylene Chloride | ✓ | | ✓ (b) |
| Benzene | ✓ | | |
| Toluene | ✓ | | |
| Ethylbenzene | ✓ | | |
| Xylenes, Total | ✓ | | |
| Methyl Ethyl Ketone (MEK) | ✓ | | |
| Methyl Isobutyl Ketone (MIBK) | ✓ | | |

(a) May be transformed abiotically under anaerobic conditions.
(b) May be degraded directly under anaerobic conditions.

Depending on the specific GCW configuration, dissolved and sorbed contaminants in both the saturated and vadose zones can be targeted for remediation. Free-phase LNAPLs (including fuel hydrocarbons) whose densities are less than water generally float at the top of the water table. They act as reservoirs for groundwater contamination and must be removed before using GCWs to prevent the risk of spreading the contaminants throughout the GCW treatment volume. Smearing free-phase LNAPLs would result in a more difficult problem to remediate. Free-phase LNAPLs are more easily removed by bioslurping or skimming techniques, while in contrast, DNAPLs (including most chlorinated solvents) do not float at the top of the water table because their densities are greater than water and they are much more difficult to remove than LNAPLs. The presence of DNAPLs will prolong the time required to remediate a site, because they will greatly increase the overall mass of contaminants that must be removed and, like LNAPLs, they will act as sources for groundwater-dissolved contamination. While it may not be possible or cost-effective to remove free-phase DNAPLs before remediating the site, every effort must be made to prevent the spreading of DNAPLs during operation of any remediation system.

In the saturated zone, contaminants generally either partition (i.e., adsorb) to the solid phase or remain in the aqueous phase. Their solubilities determine their maximum concentrations in the aqueous phase, while their actual concentrations over time depend on the extent of adsorption onto the aquifer solid phase. The adsorption potential is a function of the chemical characteristics of the contaminant and the physical properties of the soil.

Because of their nonpolar (i.e., hydrophobic) nature, sorption of fuel or chlorinated hydrocarbons usually occurs via hydrophobic bonding to organic matter (Battelle, 1995). In general, the degree of sorption is empirically related to the organic content of the soil, and the octanol-water partition coefficient ($K_{OW}$), which is a measure of the hydrophobic characteristics of the contaminant. The $K_{OW}$ is the ratio of equilibrium concentrations of a contaminant in octanol and water. Higher soil organic content and/or higher $K_{OW}$ values result in increased adsorption that retards the mobility of the contaminants in the groundwater. GCWs designed to mobilize the contaminants from the aquifer to the GCW require the contaminants to be in the dissolved phase, and adsorbed contaminants will have to desorb into the groundwater before they can be transported to the GCW.

Solubilities of some common environmental contaminants and their $K_{OW}$ values are shown in Table 7, along with their Henry's constants. Contaminants such as *n*-hexane and naphthalene are expected to be relatively immobile in the environment due to their relatively low solubilities and high $K_{OW}$ values. Thus, GCWs should not be expected to mobilize those contaminants via groundwater recirculation. In contrast, compounds such as benzene, TCE, and TCA have higher aqueous solubilities and lower $K_{OW}$ values, and they should be more readily transported to the GCW.

**Table 7. Solubilities, Henry's Constant, and $K_{ow}$ Values for Some Common Organic Contaminants.**

| Compound | Solubility (g/L) | H (atm m$^3$/mol) | $K_{ow}$ (mL water/mL octanol) |
|---|---|---|---|
| Trichloroethylene | 1.10 | 9.10E-03 | 195 |
| Tetrachloroethylene | 0.15 | 2.59E-02 | 398 |
| 1,1,1-Trichloroethane | 4.40 | 4.92E-03 | 309 |
| 1,1,2-Trichloroethane | 4.50 | 1.17E-03 | 295 |
| Benzene | 1.77 | 5.40E-03 | 135 |
| Toluene | 0.53 | 6.37E-03 | 537 |
| Ethylbenzene | 0.16 | 8.39E-03 | 1,413 |
| Xylenes | .16-.18 | 7.04E-03 | 1,318-1,585 |
| *n*-Hexane | .011 | | 10,000 |
| Naphthalene | 0.00003 | 4.84E-4 | 2,239 |

### 1.4.2 Site Hydrogeology

The hydrogeology of both vadose zone and saturated zone are important factors that govern the implementation of the GCW technology. The occurrence and movement of groundwater is a function of the characteristics of the geologic formation that hosts the aquifer. These characteristics often vary over short distances both vertically and horizontally. The geologic variables that have the most influence on the hydraulic properties of an aquifer include the rock or sediment type, facies changes, stratigraphy, type and degree of mineralization, structural features, and weathering. The geology of the vadose zone is especially important for GCW applications employing configurations that incorporate SVE, vadose zone treatment of off-gas, or infiltration galleries for recharge of treated water. The main variables effecting these configurations include soil-gas permeability, biodegradation capacity, soil moisture, and saturated hydraulic conductivity.

A preliminary characterization of the site geology is necessary to identify formation characteristics that may affect groundwater flow. A search for background geologic information must be completed as part of this characterization. This could include a literature search, and a review of maps and aerial photographs. The US Geological Survey, state geological surveys, geological societies, agricultural organizations (such as the US Department of Agriculture), environmental protection agencies, and local universities are likely sources of both regional and local geological information. The property owner, state highway departments, regional or local development authorities, and commercial aerial photography, surveying and mapping services should be contacted for aerial photographs of the site. These photographs can be used to evaluate surface water features, man-made drainage networks, irrigation systems, and structural features which may affect groundwater movement. Water well construction and abandonment records, oil and gas well records, surface mining permits, and geotechnical borings should be identified and reviewed. These records may be available from state geological surveys and local or county health departments. Geotechnical records and well logs may be available from the property owner for on site construction, and production or monitoring wells.

A site walk over should also be completed. At this time, surface features identified in the geological information, maps, and photographs can be verified. Existing wells and structures can be identified and potential locations for the circulating well(s) and monitoring wells can be evaluated. Areas of exposed geologic strata at and adjacent to the site should be inspected, if accessible.

The geological background and site information should be assembled and a conceptual model of the subsurface constructed. The model can be used as a basis for further exploration and development of the vadose and saturated zones.

Monitoring wells, soil borings, and production well borings can be installed at the site to characterize the saturated zone. Intact formation samples (cores, split-spoons, thin-walled tubes) must be collected to provide a field description of the geological conditions and to identify or estimate hydrogeological properties of the aquifer. Samples can be collected to chemically characterize the saturated soils and to measure the physical and geochemical properties of the aquifer. Groundwater samples can be collected to characterize the nature and extent of contamination and general groundwater quality.

**1.4.2.1 Saturated Zone.** The physical characteristics of an aquifer and its ability to transmit water will control the ability of a GCW to collect and disburse groundwater. Well yield and efficiency can be enhanced through proper design and construction. But ultimately, well production is dependent on physical properties of the aquifer and its host formation. Both conceptual and mathematical hydrogeologic models, which can be used to design and locate groundwater wells, assume an aquifer of uniform properties and infinite lateral extent. In nature, these conditions never occur. The following discussion identifies some of the variable characteristics of geologic formations and aquifers that exercise the most control over groundwater flow and which are considered in selection and design of GCWs.

**1.4.2.1.1 Porosity.** Porosity is defined as the ratio of the void spaces in a unit of soil or rock to the total volume of that unit. Porosity is usually expressed either as a percent or in decimal fraction (Freeze and Cherry, 1979). Table 8 lists representative ranges of porosity.

**Table 8. Ranges of Porosity for Various Soil and Rock Types (Freeze and Cherry, 1979).**

| Material | *n* (%) |
|---|---|
| **Unconsolidated deposits** | |
| Gravel | 25-40 |
| Sand | 25-50 |
| Silt | 35-50 |
| Clay | 40-70 |
| **Rocks** | |
| Fractured basalt | 5-50 |
| Karst limestone | 5-50 |
| Sandstone | 5-30 |
| Limestone, dolomite | 0-20 |
| Shale | 0-10 |
| Fractured crystalline rock | 0-10 |
| Dense crystalline rock | 0-5 |

The porosity of a sample of lithified aquifer material can be determined relatively easily in the laboratory. This is done by drying the sample to remove any moisture clinging to the surfaces in the sample, except for water, which is hydrated as a part of certain minerals. The dried sample is then submerged in a known volume of water and allowed to remain in a sealed chamber until it is saturated. The volume of voids is equal to the original water volume less the volume in the chamber after the saturated sample is removed. This method excludes pores not large enough to contain water molecules and those which are not interconnected.

The total porosity can be estimated from Equation 2.

$$n = 100\left[1 - \left(\frac{P_b}{P_d}\right)\right] \qquad (2)$$

where: $n$ = the porosity (percentage)
$P_b$ = the bulk density of the aquifer material (g/cm$^3$)
$P_d$ = the particle density of the aquifer material (g/cm$^3$)

The bulk density of the aquifer material is the mass of the sample after drying divided by the original sample volume. The particle density is the oven-dried mass divided by the volume of the mineral matter in the sample as determined by the water-displacement test (Fetter, 1994).

Determination of the porosity of aquifer sediments is not as straightforward because the porosity is dependent not only on the gain-size distribution, but the packing, or arrangement of the sediment particles relative to one another.

**1.4.2.1.2 Stratigraphy.** Stratigraphic layers have a distinctive aspect or appearance as a result of being deposited or formed under particular conditions (Sanders, 1981). Facies changes refer to both gradual and abrupt transitions of matrix within a sedimentary or unconsolidated formation and between formations. These transitions are related to the original depositional environment of the formation(s) and result in contrasting zones of hydraulic properties in both horizontal and vertical directions. Facies changes can occur at a relatively small scale, within the length of a well screen and within the ROI of a well.

Stratigraphic variations conventionally refer to transitions in the vertical direction. Stratigraphy must be considered in the design and installation of GCWs to prevent cross contamination within or across the screened zones of two hydraulically distinct water bearing zones.

**1.4.2.1.3 Hydraulic Conductivity.** The hydraulic conductivity of an aquifer is a measure of its ability to accommodate water flow and is expressed as the rate at which water can move through a permeable medium. Conceptually, the hydraulic conductivity of an aquifer is the volumetric flowrate which the aquifer will permit through a unit surface area and motivated by a unit hydraulic gradient. The units used to describe hydraulic gradient are derived from units of volumetric flowrate normalized to surface area, or $(L^3/t)/L^2$, which reduces to L/t. Common specific units for hydraulic conductivity are cm/s and ft/d.

Darcy's law concisely describes the relationship among the flowrate that a defined surface area of aquifer will accommodate under a specific hydraulic gradient and can be expressed using Equation 2 as follows:

$$Q = K\frac{dh}{dl}A \tag{2}$$

where: Q = volumetric flowrate, $cm^3/s$
K = hydraulic conductivity, cm/s
dh/dl = hydraulic gradient, unitless
A = cross-sectional area of flow, $cm^2$.

Hydraulic conductivity is perhaps the most important aquifer parameter governing fluid flow in the subsurface. The velocity of groundwater movement and dissolved contaminant migration are directly related to the hydraulic conductivity in the saturated zone. The removal efficiency of GCWs is dependent on both of these factors. In addition, subsurface variations in hydraulic conductivity directly influence contaminant fate and transport by providing preferential pathways for contaminant migration.

Because GCWs incorporate a horizontal and vertical flow component, both the horizontal and vertical hydraulic conductivities impact the system performance. Both of these parameters must be known with some precision to effectively model and design a GCW system. While estimates of hydraulic conductivity are commonly used to determine likely flow velocities and travel times for contaminants and groundwater, estimates are not sufficient for designing GCW systems.

The most common methods used to quantify hydraulic conductivity are single- and multiple-well pumping tests and slug tests. Both of these test methods have the common disadvantage that the resulting hydraulic conductivity values represent the "average" hydraulic conductivity over the length of the well screen in the test well(s) used to perform the tests. However, there are no feasible field tests currently available that do not share that disadvantage. One test that is more appropriate for GCW applications is the dipole test described in section II.

Laboratory tests on core samples provide information on the hydraulic conductivity of various layers along the screen length. Unfortunately, these analyses have three distinct disadvantages. First, cores are a very small sample and may not adequately represent the aquifer layer. A second disadvantage is cores are disturbed and often compacted upon sample collection. Finally, it is difficult to properly scale the laboratory test to the dimensions that will accurately reflect the conditions of the actual well and aquifer interactions.

### 1.5. Design Constraints of Groundwater Circulating Wells

**1.5.1 Area of Influence.** The area of influence of a GCW can be defined as the horizontal distance from the well to the farthest point at which circulation flow is still significant (Herrling et al., 1991). The area of influence is dependent on the geology/hydrogeology of the site as well as the design of the GCW itself.

Herrling et al (1991) reported that in an absence of natural groundwater flow, the sphere of influence (R) of a GCW is dependent on the:

- anisotropy (horizontal over vertical hydraulic conductivity: $K_h/K_v$)
- thickness of the aquifer
- length of the screen sections at the top and bottom of the aquifer.

In addition, the separation distance between the screen sections will affect the area of influence. In the presence of natural groundwater flow, the sphere of influence is defined by a stagnation point that occurs between the circulation well-induced flow and the ambient groundwater flow.

The vertical depth of influence of a GCW is dependent on the construction of the well, on the penetration of the well (partially or fully penetrating relative to the aquifer thickness), on the natural groundwater flow velocities, and on the anisotropy of the aquifer materials.

The area of influence is best assessed prior to installation with the application of a three-dimensional groundwater flow model that incorporates as much site-specific information as possible. The aquifer thickness and hydraulic conductivity (both $K_h$ and $K_v$), the ambient groundwater flow velocity and direction, and the well configuration, including well screen lengths, placement, and pumping rates, must all be accounted for in the modeling effort.

Previous investigations pertaining to the understanding and prediction of hydraulic heads and groundwater flow near a GCW have primarily focused on the assessment of GCWs in confined aquifer settings. However, most GCW installations have been performed in aquifers under unconfined conditions. Herrling et al (1991) reported results of a three-dimensional groundwater flow modeling effort that investigated both fully (perfectly) and partially (imperfectly) penetrating wells, both under confined conditions. Philip and Walter (1992) described a semi-analytical technique for predicting the steady-state hydraulic head and flow fields caused by the operation of multiple vertical circulation wells in a confined aquifer with a regional gradient. Stallard et al. (1996) reported results of laboratory-scale model aquifers and two-dimensional numerical modeling of GCW systems for partially-penetrating wells under unconfined conditions. This investigation used tracer studies with a laboratory-scale tank and constructed a two-dimensional flow and transport model. The numerical models were determined to be effective in estimating the plume shapes and capture efficiencies in the investigated two-dimensional case.

The area of influence of a GCW plays a crucial role in the determination of well placement and the design of a treatment system or network of GCWs. Groundwater modeling must be performed to estimate the area of influence prior to installation of a GCW. Following installation, determination of the actual area of influence can be made in the field through the use of monitoring wells/piezometers, water-level measurements, downhole or in situ flow meters, and tracer tests. It is recommended that a GCW network be initiated with one well that can be monitored to validate the model and verify the area of influence. Once the model calculations are validated, installation of the remaining wells in the network can proceed. Validating the design specifications with a single well will ensure that the network design and well spacing are adequate to meet the remediation objectives.

### 1.5.2 Treatment Limitations

As with any technology, GCW systems have treatment limitations that restrict their use for groundwater remediation. GCWs are designed to circulate water within an aquifer. The circulated water is expected to transport the contaminants from the aquifer to the GCW for treatment in the GCW unit, and/or to transport dissolved oxygen (DO) or nutrients to the aquifer to promote in situ contaminant degradation. The ability to transport the contaminants to the GCW, or to transport DO or nutrients to the area of contamination depends on the following factors:

- The amount of water that is circulated (i.e., pumped) by the GCW
- The subsurface hydrogeology
- The nature and extent of contamination
- The physical/chemical characteristics of the contaminant.

**1.5.2.1 Groundwater Pumping Rates, and Subsurface Geology and Hydrogeology.** The ability to transport contaminants from the aquifer to the GCW, and to transport nutrients into the aquifer, or both, will depend both on the amount of, and the rate at which, water can be circulated by the GCW, and on the physical characteristics of the aquifer (i.e., the permeability, hydraulic conductivity, and heterogeneity). The GCW pumping rate will determine the groundwater recirculation rate, and thereby the rate of mass transfer to and from the GCW. The hydrogeologic characteristics will determine the ROI of the GCW and the groundwater recirculation flow characteristics, or flow lines. It is important to recognize the treatment limitations that physical and hydrogeological conditions can impose on GCWs, including the following:

- Hydraulic conductivity determined in both the horizontal ($K_h$) and vertical ($K_v$) directions
  - Impermeable soils will result in slow, restricted groundwater recirculation, while highly permeable soils may result in short circuiting
  - Anisotropic soils where $K_h > K_v$ are desirable to promote horizontal groundwater flow
  - Impermeable layers (aquitards) between the upper and lower well screens could hinder or prevent the flow of groundwater between the upper and lower wells

- The depth to water table (i.e., the vadose zone depth)
  - Determine whether the vadose zone can be used for GCW off-gas remediation
  - Off-gas remediation in the vadose zone requires a sufficient residence time; deeper vadose zones will result in longer residence times of the GCW off-gas for biological degradation of off-gas contaminants

- The submergence (the ratio of the well depth below the water table to the total depth below the ground surface) must meet the minimum requirements to ensure adequate groundwater circulation

- GCW groundwater recirculation flow rates must overcome regional groundwater flows.

Each of these parameters, the horizontal and vertical hydraulic conductivities, the vadose zone depth , the submergence, and the GCW pump rate will influence the transmissive capacity of the aquifer, and the quantity of water that can be moved through the aquifer using a GCW. Thus, they must be determined on a site-by-site, case-by-case basis.

**1.5.2.2 Soil Geochemistry.** Soil geochemistry will affect the long-term operation and success of GCWs, depending on the physical, chemical, or biological processes being employed. The water entering a GCW usually will contain very little oxygen. However, air lift pumping and/or air stripping will result in the introduction of DO in the water. Iron and manganese in the groundwater may react with oxygen to form iron or manganese oxides. Thus, the iron and manganese may exert an oxygen demand, rendering less oxygen available for microbial growth. In addition, these oxides could precipitate and clog the well screens, the surrounding soils, or the treatment processes within a GCW.

In anaerobic systems, these metals generally are more stable in their soluble forms and do not risk precipitating. If the GCW is designed to promote biological degradation of the contaminants, the engineer must ensure that the pH is close to neutral and that there is sufficient buffering capacity (carbonate alkalinity) to maintain neutral pH during biodegradation of the contaminants.

**1.5.2.3 GCW Process Limitations.** GCW treatment processes include in-well air stripping, in-well activated carbon treatment, or in situ and in-well biological treatment. Each treatment process has certain advantages and limitations. In-well air stripping has the advantage of being employable with a variety of well sizes. Thus, relatively small, inexpensive GCWs can be used. However, it is limited by the volatility of the contaminants and the aeration capacity of the

GCW. Air stripping is most efficient when fine bubbles are used, while air lift pumping is most efficient when larger diameter bubbles are used. The use of larger air bubbles to meet the need of the air lift pump reduces the air stripping capacity, and a larger well size may be required to meet the air stripping requirements. In addition, the contaminated air stream must be treated with GAC or alternative methods, or be injected into the vadose zone where the contaminants can be biologically degraded before being released into the atmosphere. GAC treatment of the off-gas usually is conducted aboveground and adds to the capital and operating costs of the GCW process. In situ treatment in the vadose zone requires (1) that the contaminant be aerobically biodegradable and (2) that there is sufficient depth in the vadose zone to provide the required residence time.

In-well GAC treatment generally requires large GCW well diameters to meet the GAC requirements of the contaminated groundwater. The GAC must be able to be easily removed and replaced. GAC disposal and replacement will add to the long-term operating costs of a GCW. Aboveground GAC treatment is possible and permits the use of a smaller GCW diameter. However, aboveground treatment implies that the extracted groundwater will breach the ground surface, and will be reinjected after treatment, which may result in more stringent regulatory requirements than in-well treatment options.

For in situ aerobic biological treatment, the contaminants must be able to be degraded aerobically and the GCW must meet the oxygen demand of the contaminants in the aquifer. Table 9 shows the theoretical oxygen demand for one pound each of benzene, toluene, ethylbenzene, xylenes, or hexane. The liquid volume required to degrade 1 lb hydrocarbon assumes that the water is saturated at 8 mg/L DO. The theoretical remediated soil volume assumes that the groundwater is saturated with each contaminant and that there are no physical or chemical oxygen demands exerted by the soil, such as for iron or manganese oxidation, nitrate oxidation, or sulfate oxidation. The actual oxygen demand will depend on the geochemistry of the aquifer, and the contaminant.

**Table 9 Theoretical Oxygen Pumping Requirements for Biodegradation of Some Hydrocarbon Contaminants.**

| Compound | Oxygen Demand lbs (kg) | Liquid Volume to Degrade 1 lb Hydrocarbon[(a)] | Theoretical Remediated Soil Volume, Assuming Contaminant-Saturated Conditions[(b)] ($ft^3$) |
|---|---|---|---|
| Benzene | 1.4 (3.08) | 46,100 gal (174,600 L) | 30 |
| Toluene | 1.4 (3.13) | 47,000 gal (177,700 L) | 100 |
| Ethylbenzene | 1.4 (3.17) | 47,500 gal (180,000 L) | 334 |
| Xylenes | 1.4 (3.17) | 47,500 gal (180,000 L) | 334 |
| Hexane | 1.6 (3.53) | 53,000 gal (200,600 L) | 485 |

(a) Assumes air-saturated $H_2O$; 8 mg/L dissolved oxygen (DO)
(b) Assumes the aqueous-phase is saturated with the organic contaminant, assumes 30% porosity, assumes that there are no additional chemical oxygen demands in the groundwater.

**1.5.3 Air lift Capacity vs. In-Well Air Stripping.** In GCW designs that use airlift to pump water and air stripping as the primary treatment technology, it is necessary to optimize the interaction of these two mechanisms to achieve the system's objectives. The optimum air injection rate for airlift pumping is unlikely to be the optimum rate for air stripping. Air stripping efficiency is maximized with air and water interactions requiring high specific surface areas (surface area per unit volume) as described by the dual film theory (Bird et al., 1960). Airlift pumping efficiency is affected more by the submergence depth of the air injection point. While the efficiencies of both processes are dependent on the air-to-water flow rate ratio ($Q_a/Q_w$), different mechanisms occurring at the air-water interface impact each process. Inertial interactions between the two fluids drive airlift pumping, whereas diffusive mass transfer interactions control air stripping. It must be noted that diffusive mass transfer occurs at different rates with different contaminants, while airlift pumping only weakly depends on contaminant type.

It is the responsibility of the GCW system designers to ensure that the system is constructed so as to make efficient use of energy for both air stripping and airlift pumping in coupled systems. An engineering decision may be made to divide pumping and treatment processes if the respective process optima require widely different designs or air injection rates. For example, at very deep sites the air injection rate required for pumping may be many times greater than the rate required for efficient stripping. In such a case, it may be beneficial to use a mechanical pump to lift the water to a smaller, in-well air stripping system. Vendor-supplied systems currently exist for such applications (see Appendix).

**1.5.4 Short-Circuiting of Groundwater Flow.** The potential for the short-circuiting of groundwater flow, or direct flow from the portion of the GCW with relatively higher hydraulic head to the portion of the GCW with relatively lower hydraulic head without flowing radially away from the well, is a potential physical constraint of GCW design. Short-circuiting can occur within an improperly grouted or packed borehole or in the immediate vicinity of the well.

Well construction and adequate grouting in the borehole are essential to the physical and hydraulic separation of the injection and withdrawal portions of the well. A continuous sand pack in the borehole would provide no physical or hydraulic separation between these two areas, allowing water to be transmitted within the borehole without traveling through the aquifer matrix. Short-circuiting may also occur due to isotropic conditions in the aquifer, in which the vertical permeability of the aquifer materials is the same as the horizontal permeability. In this situation, the driving force for horizontal flow away from the GCW is limited and flow is likely to proceed directly from the area, or screen, of injection to the area, or screen, of withdrawal.

In a study at Tyndall AFB designed to evaluate the feasibility of coupling the GCW technology with bioventing, Battelle (1995) reported a communication time of less than 2 minutes between the upper and lower screens in a system with 5 ft separation between the upper and lower screens.

Monitoring wells, soil borings, and production well borings can be installed at the site to characterize the saturated zone. Intact formation samples (cores, split-spoons, thin-walled tubes) should be collected to provide a field description of the geological conditions and to identify or estimate hydrogeological properties of the aquifer. Samples can be collected to chemically characterize the saturated soils and to measure the physical and geochemical properties of the aquifer. Groundwater samples can be collected to characterize the nature and extent of contamination and general groundwater quality.

## 2.0 Technology Assessment And Maturity

Groundwater circulating wells have been successfully employed at numerous sites in the U.S. and Europe, and the GCW vendors have provided case history information on some of these installations (see Appendix). Because it was not possible to obtain all of the data generated on these efforts and make independent evaluations, six DoD installations were evaluated and the results and lessons learned are summarized in the following sections. These case histories should be reviewed to get a better idea of the current status of the GCW technology.

### 2.1 Current Status of the Technology

At its current level of development, the GCW technology can be classified as innovative. Although the technology has been applied at numerous sites, technology has not benefited from a DoD-funded initiative such as the bioventing, bioslurping and natural attenuation technologies had during their developmental phases. This is because GCWs are a vendor-based technology involving a number of patents. As a result, the technology needs further development and refinement before it can be classified as mature.

Due to private industry's motivation to sell their technology, data on failures or lessons learned are not forthcoming. Only too often GCW systems have been promoted as a "silver bullet" solution when in actuality other technologies may have been more appropriate. Therefore it is strongly recommended that anyone considering using GCW technology carefully read this protocol and scrutinize the technology for any specific applications. If, for any reason, the user is not comfortable with the selection of GCW technology for their site they should seek independent consultation on the technologies of applicability and/or give serious consideration to an alternative remedial technology.

Following the summaries of these demonstrations a technology assessment is provided to summarize the results and lessons learned. Additional case history information has been provided by the GCW vendors and is presented in Appendix A. This information has been provided on a voluntary basis and has not been verified for inclusion in the protocol.

### 2.2 DoD Experience with the GCW Technology

At the time this protocol was written, the DoD had sponsored or hosted several demonstrations of the GCW technology. Performance data for six of these demonstrations was reviewed and a summary of each demonstration is provided in the following sections.

**2.2.1 Port Hueneme Naval Exchange Site.** In January 1995, the Strategic Environmental Research and Development Program (SERDP) supported a joint effort between the U.S. Naval Research Laboratory (NRL), the U.S. Environmental Protection Agency (EPA), Texas A&M University, SBP Technologies, Inc., IEG Technologies, Inc., and Beazer East, Inc. The objective of the effort was to determine the catabolic activity of indigenous soil microflora on the constituents of interest. The strategy included The GCW technology was selected as the treatment technology for which to achieve the project objective. The following is a brief description of the GCW installation at Port Hueneme Naval Exchange site (NEX) and the findings that were relevant to implementing the GCW technology. More detailed information can be found in Spargo, 1996.

The site was contaminated with approximately 11,000 gallons of gasoline that leaked from two delivery lines between September 1984 and March 1985. The soils within 10 m of the surface at the site consist of three units: (1) fine-grained silty sand to 1.7 m below ground surface (bgs), (2) fine- to coarse-grained sand to approximately 6.2 m bgs, and (3) sandy-to-silty clay unit encountered between 6.2 and 8 m bgs. There are five aquifers beneath the site; however, the contamination was confined to the upper perched aquifer and the research focussed on this formation.

The depth to water was between 1 and 3.7 m bgs. The hydraulic gradient at the site was 0.001 ft/ft to the southwest and groundwater flowed in this direction as a velocity of 231 – 548 m/yr. The horizontal hydraulic conductivity was measured at 3.84 x 10-2 cm/s and the vertical hydraulic conductivity was estimated at 3.84 x 10-3 cm/s. is aquifer was characterized with the following parameters.

The GCW system consisted of four GCW wells. One GCW-400 system was installed in a 400-mm ID. well casing (GCW-400) placed near the source. Three GCW-200 systems were installed in 200 mm ID. well casings (GCW-200) downgradient of the main spill. These wells were placed to provide plume containment.

The upper screen of the GCW-400 extended from 2.4 to 4.8 m bgs to facilitate discharge of the treated groundwater and to maximize the SVE/bioventing in the vadose zone. The lower screen extended between 6.64 and 8.01 m bgs. The well was equipped with a single three-phase, 208 volt, 5-hp blower and a single three phase, 110-volt, 0.5-hp submersible pump. Four sets of three monitoring wells (12 total) were installed around the GCW-400. The wells were installed to provide shallow (3.2 to 3.7 m bgs) and deep ( 7 to 7.2 m bgs) samples. A vadose zone sampling port was included at 2.3 to 2.5 m bgs.

The three GCW-200 wells were installed with overlapping ROIs to form a "biocurtain" to prevent the off-site migration of contaminant. The upper and lower screens were placed between 2.2 and 5 m bgs and 7 and 8.7 m bgs, respectively. Based on a calculated stagnation point distance of 18.91m and a modeled width of the circulation cell of 38.91 m, the wells were placed 40.02 m apart. A total of eight monitoring wells were placed around the biocurtain. The eight monitoring wells were configured the same as the wells around the GCW-400 system.

An intensive monitoring schedule was followed to asses the performance of the GCW systems and monitor the biodegradation of the contamination. Weekly, duplicate groundwater samples were collected from selected well and analyzed for pH, dissolved oxygen (DO), inorganic nutrients (N,P, fhemical oxygen demand (COD)), and temperature. Duplicate soil-gas samples were collected from each of the 20 monitoring wells on a regular basis and analyzed for oxygen, carbon dioxide, and methane concentrations and stable isotope ratio. In addition to the more routine sampling above, eight quarterly sampling events were conducted rigorous detailed analysis of system performance (see Spargo 1996 for details). Dye tracer tests were conducted to verify groundwater circulation.

The data showed that within 8 months of operation, BTEX concentrations in the shallow monitoring wells closest to the GCW-400 system were reduced by 52% decreasing from 4.66 mg/L to 2.88 mg/L. It was concluded that free product in the vicinity of the GCW-400 system was responsible for the low removal performance. Data from the biocurtain wells showed the BTEX concentrations in the shallow well in the center of the biocurtain were reduced by 97%

over the first three months of operation decreasing from 77s mg/L to < 26 mg/L. After 6 months, BTEX concentrations in samples from these wells were below 0.002 mg/L which was in compliance with the California groundwater drinking criteria. BTEX concentrations in groundwater from the deep wells were reduced from 118 mg/L to under 0.001 mg/L after 6 months of system operation. Based on these results, it was concluded that the biocurtain was effective in providing in situ containment.

Although the results presented in the above referenced report appear promising, too little data were presented in the report to make any conclusions about the design and operational performance of the GCW at Port Hueneme. The final monitoring data analyses along with the results from the tracer studies will be presented in a forthcoming report. It is strongly recommended that the reader review these additional results before making any conclusions on the performance of the GCW systems at Port Hueneme.

**2.2.2 Hill Air Force Base.** In 1996, the Air Force sponsored a 44-week technology demonstration of the UVB technology at OU 6 at Hill AFB, Utah beginning in January and ending in November. The primary contaminant at the site was TCE. The objectives included: (1) determine the ability of the UVB technology to reduce TCE concentrations in the aquifer to 5 µg/L; (2) to determine what parameters are most useful for monitoring the system; and (3) to evaluate the feasibility of going full-scale with the technology at the demonstration site. The following sections provide a summary of the test and results. More detailed information can be found in the report *Technology Performance and Application Analysis of UVB Groundwater Circulating Well Technology, Operable Unit 6, Hill AFB, Utah* (Radian, 1996) and *ATTACHMENT to the Technology Performance and Analysis of UVB Groundwater Circulating Well Technology, Operable Unit 6, Hill AFB, Utah (Radian, 1997).*

OU 6 is located in the northern part of Hill AFB in an area that was used for, or in support of, maintenance and testing operations. It is believed that operations related solvent use began around 1960. TCE is the primary contaminant of concern at OU 6 although other contaminants have been detected at below their respective maximum contaminant levels (MCLs). It is believed that these contaminants have been introduced into the environment through leaking underground storage tanks and possibly surface spills. The resulting plume was approximately 500 x 3,000 ft with a maximum detected TCE concentration of 440 µg/L. The average peak concentration in the center of the plume was typically between 200 and 300 µg/L.

The geology at the site consists of fluvial-deltaic deposits. There are three aquifers underlying OU 6 with the contamination confined in the upper aquifer which is between approximately 105 and 135 feet bgs. The aquifer is predominantly fine-grained sand with some variability in grain size and density distribution. A fairly continuous, 0.1 to 3.0 ft thick clayey silt layer exists at approximately 110 feet bgs. The hydraulic gradient was 0.02 ft/ft with groundwater flow to the north at approximately 0.53 to 1.8 ft/day.

A constant rate aquifer pump test was conducted to characterize the aquifer in the test area. Three horizons were monitored during this test. The results showed that the aquifer response was similar among the horizons, indicating that there was communication between the three horizons. The horizontal hydraulic conductivity ranged between $2.6 \times 10^{-3}$ and $9.6 \times 10^{-3}$. The storativity ranged between 0.0004 and 0.029.

The test system at OU 6 consisted on one UVB 200 system, three 2-inch diameter annulus wells, and ten monitoring well clusters. The UVB system consisted of an 8-inch diameter steel casing with an 8-ft upper screen section that straddled the water table, and a 4-ft screened section that was placed between 127 and 131 ft bgs. The system was designed to provide an air-to-water ration of 50:1, a stripping efficiency of 90 to 99%, an airflow rate of 60 scfm, and a groundwater throughput of 8.8 gpm. Based on modeling by the vendor, the theoretical capture zone width of the top and bottom of the zone was expected to be 16 and 175 ft, respectively. Nine of the ten monitoring well clusters were placed within this zone of influence. Each monitoring well cluster consisted of three monitor wells containing 5-ft long screened sections placed at three depths to cover the thickness of the aquifer. The screened sections were isolated from each other by a bentonite seal. The annulus wells installed in the borehole for the UVB system and contained 5-ft screened sections placed adjacent to the UVB well screens. Two annulus wells were placed adjacent to the upper screen and one adjacent to the lower screen. These wells were used to monitor the influent and effluent from the UVB system.

The UVB system was started in January 1996 and operated for 44 weeks. During this time, TCE concentrations were measured in the aquifer and around the well. Bromide tracer tests were conduced to determine the groundwater throughput in the well. This entailed injecting a known concentration of tracer at a known flow rate into the influent section of the well and monitoring the dilution of the tracer at the effluent section of the well. A divergent tracer test was conducted using fluorescein dye to determine the ROI of the well. This entailed injecting the dye into the air intake line and monitoring for the appearance of the dye in the monitoring wells surrounding the UVB well. A converging tracer test was conducted using sulfur hexafluoride ($SF_6$) to determine if groundwater from a distinct point within the design ROI would be captured by the UVB well. This test entailed injecting $SF_6$ into one of the mid-level monitoring wells and monitoring for its appearance at the UVB well.

Four major system configuration changes were made during the 44-week operation period based on observations made in the field. The first change was the installation of a canister-type stripper. This change was necessary due to a rise in the static water level. The canister stripper was installed above the upper screen. The second modification was replacing the inflatable packer with a fixed packer. This was necessary because excessive leakage across the inflatable packer prevented groundwater circulation. Leakage was confirmed by the first throughput tracer test. The third modification was a reduction in the pumping rate from 17.6 gpm to the design flow of 8.8 gpm. This was done because the vendor believed that the pumping and discharge rates were more than the aquifer could handle. The final modification entailed raising the canister stripper an additional 2 feet. This was necessary because the water level had risen to the point that it was interfering with airflow and treatment at the well.

The results from the both the TCE analysis and the tracer tests showed that the UVB system did not operated according to design. As mentioned above, the first throughput tracer test indicated that the flow through the well was much lower than the 17.6 gpm rate that the submersible pump was pumping. This indicated that significant leakage across the packer was occurring. Upon replacing the packer, a second through put tracer test indicated that there was no net movement of water through the well. The results from the TCE analysis, pressure transducer readings within the ROI, and the other two tracer tests confirmed this observation.

Prior to changing the packer, there was an estimated 1 gpm throughput in the well. Although this was far below the 8.8 gpm design flow, pressure transducers in two shallow monitoring wells as far away as 30 showed some response to UVB operation indicating that the ROI at the top of the aquifer extended at least this far from the well. Based on the system throughput and an estimated groundwater specific discharge, the maximum width of the capture zone was 21 ft, approximately 20% of the average design distance. Unfortunately, the TCE concentration in the UVB influent rapidly decreased from 45 µg/L to non-detect levels and the effluent TCE concentrations remained at non-detect levels during the test period. This indicated that there was serious short circuiting in the immediate vicinity of the well and that no appreciable level of treatment was occurring at any appreciable distance from the well. This was confirmed by the lack of any decrease in the TCE concentrations in any of the monitoring well.

After the packer was changed, there was no response registered by the pressure transducers when the UVB was operated and there was no decrease in TCE concentrations is any of the monitoring wells over the duration of the test period. Other water quality parameters measured during the test also showed no indication that groundwater was being circulated. The data from the divergent tracer test showed a much higher concentration of tracer at the influent screen and a much lower than expected concentration at the influent screen. This strongly suggested that short circuiting was occurring within the well.. The results from the convergent tracer test showed that there was no communication between the mid-level monitoring well located approximately 25 ft crossgradient to the UVB well but there was communication in a well located approximately 30 feet downgradient from the UVB well. This showed that water was moving past the UVB well but not being circulated by the UVB well.

Additional aquifer tests were conducted to explain the lack of groundwater circulation. Falling-head permeameter tests revealed that the vertical hydraulic conductivity varied by more than three orders of magnitude in a core taken 30 ft from the UVB well. The data showed that the hydraulic conductivity at the UVB discharge zone was lower than the hydraulic conductivity at the UVB intake zone.

Mass balance analyses showed that the UVB system removed on average 96% of the TCE that entered the well. This was within the design stripping efficiency of 90 to 99% removal, indicating that the air stripping component of the UVB well performed as expected. The total mass of TCE removed over the 44 week period was estimated at 0.04 lbs. Based on the mass balance analysis, the TCE concentration in the off-gas should have been approximately 13 ppbv; however, TCE was only detected in one off-gas sample at a concentration of 2.2 ppbv. This indicated that the throughput in the well was even less than the estimated 1 gpm.

A cost analysis was performed for implementing and evaluating the UVB technology based on contractor estimates and actual costs from the demonstration at OU 6. The cost breakdown was as follows.

| Item | Cost |
|---|---|
| Well Components | $ 61,860 |
| Well Installation | $ 38.185 |
| Energy (1 yr) | $ 2,439 |
| Tracer Testing | $ 30,725 |
| Total | $ 133,209 |

The above analysis gives an approximation of the costs for implementing and evaluating a UVB system similar to the one at OU 6. There are however additional costs such as those associated with planning, connection to power, additional construction and site work, instrumentation, sampling and analysis, and reporting. One of the advantages that may be realized with groundwater circulating wells is a cost savings because of a smaller energy requirement without having to pump groundwater to the surface. A cost analysis of the energy costs for 1 year operation of the UVB, pump and treat, and in situ air sparging technologies showed the energy costs were $2,439, $686, and $6,859 for these technologies, respectively. This indicated that the energy savings would not be realized at the OU 6 site.

The primary conclusion from this technology demonstration was that the technology did not provide a barrier to TCE migration at OU 6. Although the system did appear to circulate groundwater and did reduce TCE concentrations across the well before the packer was replaced, the system operated far below the design specifications. After the packer was replaced, the system simply did not work. This fact as wells as the data from the tracer tests and subsequent aquifer tests point out several problems with this specific installation.

First, the hydrogeologic investigation and the data set used in designing the well was insufficient. Although the data from conventional aquifer testing indicated ideal conditions for conventional groundwater pumping, data was not collected to adequately characterize the aquifer and properly design a GCW system. Effective GCW operation relies on a vertical component of groundwater flow to drive circulation in the aquifer formation. At a minimum, it is necessary to know the vertical hydraulic conductivity to effectively predict aquifer response to GCW operation. This parameter was not determined prior to design and installation of the UVB system at OU 6. Tests conducted subsequent to UVB installation showed the significant horizontal stratification with the vertical hydraulic conductivity highly variable. Had this data been collected prior to design, the outcome from this demonstration may have been predicted. The net result illustrates the importance for conducting aquifer tests specifically designed for GCW implementation. One such test is the dipole test described later in this protocol.

The other major problem that was apparent from this demonstration is the failure of equipment. GCW systems are more complex than conventional pump and treat and the integrity of all in-well components is essential for proper operation. Although the submersible pump and in-well air stripper functioned according to design, the packers leaked resulting poor groundwater circulation. The results from the tracer tests indicated that the packers used in this system were not properly designed or that they were faulty. The head loss across the packers was apparently less than the head loss in the aquifer, causing the water to short circuit within the well. The second packer failed more than the first packer, suggesting that more consideration needs to be given to the pressure ratings of the packers during the design phase. Because the system at OU 6 was a deep system, changing the packers required heavy equipment to pull the system components from the well and the process was slow and expensive. This showed how important it is to get the design and installation of a GCW system right the first time so as to avoid the need for pulling well components for replacement and/or repair. The results from this demonstration show how important it is for the vendors to be held responsible for costs of correcting improper design or repairs due to predictable component failures.

**2.2.3 Tyndall Air Force Base.** The Air Force conducted a GCW demonstration at Tyndall AFB, Florida between July 1994 and July, 1995. The objective of this demonstration was to determine if the GCW technology could be coupled with bioventing to simultaneously treat contamination in both the saturated and unsaturated zones. The demonstration did not include extensive GCW design nor GCW system optimization. The targeted contaminants were hydrocarbons that resulted from leaking USTs holding JP-4 and diesel. Details about this demonstration can be found in Armstrong Laboratory Report AL/EQ-TR 1995-0039.

Two GCW designs were included in this demonstration. The first one, identified as a MBV system, consisted of a simple air lift pump installed in a 4-inch bioventing well that was modified to extend into the groundwater. The casing contained two screened sections, one extending between 11 and 15 ft bgs, and the other straddling the water table and between 2 and 6 ft bgs. The second well, identified as the mKGB system, was a modification of the KGB design offered by IEG Technologies, Inc. This system was installed in an 8-inch well casing that was screened the same as the other design. The monitoring system included:

- 5 piezometers with 1-ft long screens placed at the middle of the upper screen of each GCW
- 1 piezometer of similar design with the screen placed at the middle of the lower screen of each GCW
- 8 tri-level groundwater monitoring points placed varying distances from the wells with the probes located at 9,12, and 15 ft bgs.

The wells were equipped with sampling ports at the upper and lower screens of each well and a sampling port for collecting samples of the system off-gas.

The bioventing component of the system consisted of 8 dual-level soil gas monitoring points placed at the same locations as the groundwater sampling points. The probes were set at 2 and 4 ft bgs. The off gas from the GCWs was directly injected into the vadose zone to provide aeration.

The demonstration lasted for 12 months. The MBV system was operated for the first 3 months and the mKGB system was operated for the remaining 9 months. Each system was operated at an airflow rate of 1 scfm, the maximum rate that did not result in excessive discharge of contaminant vapor from the ground surface. Monitoring included:

- collection of initial soil and groundwater samples and analyses of hydrocarbons, and DO in the groundwater samples.
- a bromide tracer test was conducted to determine the ROI in the aquifer
- extensive sampling of GCW influent and effluent with analyses of hydrocarbons, and off-gas with analyses for hydrocarbons, oxygen, and carbon dioxide
- sampling and analysis of soil gas for hydrocarbons
- 5 respiration tests in the vadose zone to monitor biological activity.
- surface emission testing and analyses for hydrocarbons.

The overall conclusion from this demonstration was that these two technologies could be effectively coupled to treat hydrocarbon contamination at this site and the coupled technologies had potential for application at other sites. In addition several observations were made about the performance of the performance characteristics of the GCWs in general. First, the wells did achieve the 25 foot ROI: however, there was rapid communication between the two screens (2 minutes) whereas it took much longer (up to 3 months) for water to circulate throughout the circulation cell. This resulted in a high rate of circulation and a rapid decrease in the hydrocarbon concentration in the influent to the wells which meant that a significant portion of the energy was going to circulating clean water close to the well.

A second observation was that the air lift pump was effective at stripping contaminants from the aqueous phase and at oxygenating the water as it moved through the well. The influent DO was consistently below 1 mg/L while the effluent DO was consistently greater than 5 mg/L. Although this aeration efficiency was promising, the oxygen demand in the aquifer rapidly depleted the oxygen and no increased DO was observed in any of the monitoring points, not even five feet from the wells. Because greater than 19% of the oxygen remained in the system off-gas, and the rate of mass delivery was low, the GCW in this application was not an efficient method for supporting aerobic biodegradation in the aquifer.

An attempt was made to determine the groundwater pumping rate by conducting a mass balance on TPH and 11 molecular weight ranges of the contaminant entering and leaving the well. The following relationships were used.

$$\text{TPH Flux}_{inf} = \text{TPH Flux}_{eff} + \text{TPH Flux}_{off\text{-}gas}$$
$$\text{TPH Flux}_{inf} = \text{Water Flow Rate} \times \text{Conc}_{inf}$$
$$\text{TPH Flux}_{eff} = \text{Water Flow Rate} \times \text{Conc}_{eff}$$
$$\text{TPH Flux}_{off\text{-}gas} = \text{Air Flow Rate} \times (\text{Conc}_{off\text{-}gas} \times \text{CF})$$

where:

$\text{TPH Flux}_{inf}$ = mass of TPH entering the well in the aqueous phase per unit time
$\text{TPH Flux}_{eff}$ = mass of TPH exiting the well in the aqueous phase per unit time
$\text{TPH Flux}_{off\text{-}gas}$ = mass of TPH exiting the will in the vapor phase per unit time
Water Flow Rate = volume of water entering or leaving the well per unit time
Air Flow Rate = volume of air injected into the well per unit time
$\text{Conc}_{inf}$ = mass of contaminant per unit volume of influent water
$\text{Conc}_{eff}$ = mass of contaminant per unit volume of effluent water
$\text{Conc}_{off\text{-}gas}$ = mass of contaminant in system off gas on a ppm basis
CF = conversion factor to convert ppmv to mass per unit volume

Hydrocarbon concentrations were measured in influent, effluent, and off-gas samples. The airflow rate was set at 1 scfm. The only unknown in was the water flow rate. The results from this exercise calculated in an average flow rate of 4.53 L/min with a C.V. of 69 percent. The large variability may have been the result of sampling from a single sampling probe located at the influent screen of the GCW may not have provided a representative sample of the water entering the well. If this method is to be used to determine pumping rates, the samples must be collected from within the well after mixing of the influent water but before any air stripping or other treatment occurs. The variability in the results using a single sampling probe outside the influent screen showed that this method was not accurate enough use for monitoring pumping rates of GCWs.

**2.2.4 March Air Force Base.** In 1993 through 1994, a demonstration of the UVB technology was conducted at March Air Force Base in Riverside, California. The EPA Superfund Innovative Technology Evaluation (SITE) Program evaluated the performance of the system over the first 12-months of the demonstration. The system was operated for an additional 6 months for further evaluation. The participating vendors included IEG Technologies Corporation, Roy F. Weston, Inc., and Black & Veatch Waste Science, Inc. The primary objective was to evaluate the feasibility of using the UVB technology for removing chlorinated VOCs, primarily TCE, from the groundwater at March AFB, and to evaluate the cost effectiveness of the technology. The information presented below, as well as more information describing this demonstration, can be found in EPA SITE Programs report EPA/540/R-95/500a and Bannon et al, 1995.

The demonstration was carried out at site 31, an unclassified solvent disposal site that is within OU 1 at March AFB. In general, the geology at the site is alluvial and fluvial deposits consisting of laterally discontinuous units of fine-grained sediment dominated by fine-grained sand and silt. The depth to groundwater was approximately 40 feet and groundwater flow direction was toward the south with an average gradient of 0.007. The average hydraulic conductivity was calculated at 90.5 gpd/ft$^2$, the effective porosity at 27.2%, and the transport velocity at 0.31 ft/day.

An initial site characterization consisting of collecting on continuous core to a depth of 118.5 feet and analyzing the core for geochemical, chemical, microbiological, and lithological parameters. The results of the lithologic analysis showed that there was a zone of low permeability at approximately 85 feet bgs, so the well was designed and installed above this depth.

The system included one 16-inch diameter UVB well installed to 87.5 feet bgs with a 26-inch bucket auger. The well was operated in an upflow mode and contained a submersible pump and an air stripping chamber. The well was equipped with three 2-inch PVC monitoring wells placed in the annulus of the well borehole, one well was screened across the influent section and the two other wells were screened across the effluent section of the UVB. These wells monitored influent and effluent VOC concentrations. A total of 13 groundwater-monitoring wells were sampled: 9 within the expected zone of influence, 3 below the zone of influence, and one outside the zone of influence. Off-gas was treated aboveground using activated carbon. Off-gas samples were collected from sampling ports placed before the aboveground treatment system.

Performance monitoring included sampling and analyses of the influent to, and effluent from, the UVB well, groundwater from the 13 monitoring wells, and system off-gas. Approximately 165 sets of influent and effluent water samples were collected and analyzed for VOC, metals, minerals, and other water quality parameters. Groundwater samples were collected monthly for the first 6 months, then bimonthly for the remainder of the demonstration. These samples were monitored for VOC concentrations, dissolved oxygen, temperature, specific conductance and pH. System off-gas was monitored for VOC concentrations, linear flow velocity, vacuum/pressure, relative humidity and temperature.

The results from the groundwater sampling and analysis indicated that the TCE concentrations in the wells within the predicted capture zone decreased approximately 52 percent after 12 months and 62% after 18 months of operation. Over the 18 months, TCE concentrations decreased from an initial range between 160 and 1,000 μg/L to a range between 45 and 270 μg/L. TCE concentrations in monitoring wells screened below the capture zone all showed an increase. The TCE concentration in the one well outside of the capture zone did not change significantly during the 18-month operational period. The trends in TCE reductions in the 6 monitoring wells sampled over the 18 month duration showed a greater impact in wells closer to the UVB well. Results from $O_2$ and $CO_2$ monitoring suggested no enhancement in aerobic biological activity by the operation of the UVB.

The data trends in the TCE concentrations over time and with increasing distances from the UVB well suggest that the UVB was effective at reducing TCE concentrations and that the wells were within the circulation cell. The furthest well was located 90 feet from the UVB. The design circulation cell was 110 feet. The closest well outside of the design circulation cell was approximately 240 feet from the UVB well and showed no response to the operation of the UVB. Because there was a definite effect at 90 feet, it was assumed with some confidence that the actual diameter of the circulation cell was at least 90 feet. The SITE program evaluation did not support the use of variations of target concentrations in the monitoring wells for determining the ROI due to variables that were independent of the UVB system. Their evaluation did suggest that the data trends showed homogenization of the contaminant in the groundwater.

Several methods were used to determine the radius of the circulation cell. Groundwater modeling results indicated an 83 ft ROI which is close to the observed radius based on the TCE concentration trends. A dye tracer test resulted in recovery of dye 40 ft from the well in the downgradient direction, but no recovery was achieved at 40 ft in either the upgradient or cross gradient directions. These results indicate that the ROI may have been significantly less than the ROI indicated by the trends in contaminant reductions or the modeling effort.

The SITE Program evaluation of the well performance showed that the well achieved greater than 94 percent average removal efficiency for TCE during the first 12 months of operation. During this period, the influent concentrations ranged between 14 and 220 µg/L, with an arithmetic mean of 56 µg/L. The mean TCE concentration in the effluent over the 12 months was 3 µg/L. The 95 percent upper confidence limit concentration was 6 µg/L which translates into an 89 percent removal efficiency.

An attempt was made to estimate the mass of TCE removed by the UVB system. Calculations were made based on the water and on the gas phase concentrations. A TCE mass removal rate of 10 grams per day was calculated based on the difference between the influent and effluent concentration and using a pumping rate based on pump performance. The mass removal rate calculated based on the off-gas TCE concentrations and flow rate was 0.1 grams per day.

The large discrepancy between the TCE mass removal rates calculated on the aqueous and gaseous phase concentrations reveals two problems with the approach. First, taking one influent sample from one side of the GCW does not provide a representative sample of the water entering the well. Contaminant is never evenly distributed around a GCW. This combined with the heterogeneity within the zone of influence results in varying contributions in terms of flow and concentrations to the GCW in three dimensions. Collection of unrepresentative samples will lead to error in mass removal and stripping efficiency calculations. Multiple samples, of a sample after mixing, must be taken to accurately estimate either of these parameters.

Another concern raised by discrepancy in the TCE mass removal rates is the need to accurately determine groundwater-pumping rates. Inaccurate estimation of pumping rates would impact the calculation of mass removal rates. One of the advantages of GCW configurations that utilize mechanical pumps is that pump performance curves provide a viable method for determining pumping rate. To do this requires accurate measurements of pressures or head loss are required. Although these parameters were monitored during this demonstration, sufficient detail was not available to evaluate this method. The real challenge is to determine pumping rates in air lift GCW systems where surging and mixed fluids make traditional flow rate measurements difficult or even impossible.

System costs were estimated by the SITE program as follows:

- Capital costs for single treatment unit $180,000
- Operation and maintenance (1st year) $ 72,000
- Operation and Maintenance (subsequent years) $ 42,000

Based on these estimates, the costs for 1, 3, 5 and 10 years of operation were calculated to be $260,000, $340,000, $440,000 and $710,000, respectively. The costs for treating 1,000 gallons of water, the amount of groundwater pumped through the system, were estimated at $260, $110, $88, and $71 for 1, 3, 5, and 10 years, respectively. When evaluating these costs, it should be noted that between 60 to 90 percent of the water pumped through a GCW system is circulated water.

**2.2.5 Keesler Air Force Base.** AFCEE sponsored a demonstration of the Density Driven Convection (DDC) technology at Keesler AFB in Biloxi, Mississippi. The purpose of this demonstration was to evaluate the effectiveness of the DDC system for reducing TPH in groundwater and soil. Wasatch Engineering, Inc., is the owner of the DDC technology and they were responsible for pilot testing, scale-up, installation, operation, and monitoring and evaluating their system. More information on this effort can be found in AFCEE's project report (TBD).

The DDC technology was demonstrated at a gasoline station where underground storage tanks leaked causing soil and groundwater contamination. The soils at the site are XXXX. The depth to groundwater is approximately 5 feet bgs. The lateral extent of the contamination was approximately 150 feet by 225 feet in the soil and 275 feet by 600 feet in the groundwater. The plume has migrated to the east-northeast of the UST location.

A pilot test was conducted in late 1995 to collect required scale-up data. One DDC well was installed and operated for a period of 1 month. During operation, tests were conducted to ? Radius of circulation cell? Stripping efficiency? Groundwater pumping rate?.

The large-scale system was designed based on the pilot test data. The scaled-up system included 38 DDC wells placed in an L-shaped pattern in the source zone area along the east side and south east cornet of building 1504. The wells were constructed out of XXXXX and were screened between XX and XX feet bgs. The monitoring system included 8 piezometer pairs installed on perpendicular axis on the eastside of building 1504. The piezometers were screened between XX and XX feet bgs. Groundwater data also were collected from XXX existing monitoring wells.

Operational parameters? air to water ratio?

The system was turned on in May 1996 and operated for 356 days. During this time, eight monitoring events were conducted during which water table elevations, groundwater contaminant and DO concentrations, influent and effluent contaminant and DO concentrations, and SVE off-gas flow rate, TPH, $O_2$, and $CO_2$ concentration were monitored. The data were used to asses the performance of the system.

Stripping efficiency

Water table elevation measurements can be used as an indication of groundwater circulation. Only one of the piezometers registered a response to the operation of the DDC system. The water levels in this piezometer showed the expected pattern for downward circulation with an increase in the head in the upper flow zone and a decrease in the head in the lower flow zone. The magnitude of the head developed by DDC operation at this location averaged between 2 and 3 feet. The water table elevations in the other piezometers and monitoring wells showed no significant impact. While this data would support the conclusion that the DDC system did not support circulation of groundwater at this site. However, the time series of elevation measurements showed significant variability. It may be possible that a more sensitive method for measuring water table changes may have better shown head responses from DDC operation. One way or another, the data support the need for accurate monitoring if water table elevations are used to determine groundwater circulation.

DO data indicated that the wells were effective at oxygenating the water as it passed through the well with influent DO values in the range between 0.12 and 7.6 mg/L and the effluent DO ranging between 6.8 to 9.7. The groundwater DO indicated that even though oxygenated water was discharged from the DDC wells, the system was not effective at oxygenating the groundwater. The average DO in the well influent was 7.6 mg/L suggesting that the well had significantly cleaned the aquifer close to the well; however, DO concentrations in the aquifer remained at pretreatment levels. This data supports the conclusion of poor groundwater circulation that was based on the head changes as described above.

MORE INFORMATION TO COME, WAITING FOR FINAL REPORT.

**2.3 Technology Assessment**

Even though GCW systems have been installed at well over 100 sites throughout the United States and Europe, the technology is still in the maturation stage. The six demonstrations conducted at DoD installations described in the preceding sections have resulted in varying degrees of success, and the data illustrate some of the areas where further refinements are required before the technology can be either screened during technology selection or implemented reliably for site remediation. This section discusses the overall status of the technology taking into account the lessons learned from these six demonstrations.

One of the most obvious requirements for proper implementation of the GCW technology is the need for a detailed hydrogeologic investigation that defines the conditions at the site in 3 dimensions. Many of the shortcomings of the demonstrations described above could have been prevented had there been more focus on the up front effort. Conventional pump and treat technologies rely primarily on horizontal flow to the extraction wells or between extraction and injection wells. The effectiveness of GCW systems relies on the development of a circulation cell which relies on a vertical flow component. Thin layers of less permeable strata can interfere with circulation and significantly decrease the effectiveness of GCW systems. Conventional aquifer testing during site characterization may not reveal these strata. In addition, conventional pump tests are not designed to determine the vertical conductivity, a parameter that must be known for proper GCW design. It is imperative that both the vertical and horizontal hydraulic conductivity be determined both at the well as well as throughout the circulation cell. This data is needed for modeling the aquifer response to GCW operation to effectively place both the screen intervals and the multiple wells at any given site. There are no rules of thumb that can reliably design or place GCWs and any attempt to do so must be rejected. Although the up front site characterization will be more expensive than that required for other technologies, it is necessary for proper design and implementation.

Determining the direction of groundwater flow is necessary for the proper design of any remedial technology but it is critical for successful of the GCW technology. This is especially true for cases where the GCW technology is to be used for plume capture. Contaminated groundwater will flow past the wells if they are not designed and properly oriented to groundwater flow. The direction and flow velocity skew the circulation cell and have a direct influence on the capture efficiency. The problems encountered at both MMR and Port Hueneme illustrated the problems that will be encountered when systems are designed using the wrong flow direction.

It is apparent from the discrepancies in the results from the six demonstrations that the monitoring approaches for GCW performance need some further development. This includes methods for measuring the pumping rate, determining radius of circulation, measuring the treatment performance in the well, and measuring the treatment achieved in the groundwater.

Accurately measuring the pumping rate is required for modeling groundwater circulation and for determining in-well treatment performance. While pump performance curves can be used to determine flow rates achieved by mechanical pumps, there is a challenge in measuring the flow rate with air lift systems. There are equations that can calculate the flow rate in conventional air lift pumping; however, their use for GCW air lift systems is limited because of the different configuration for discharge from these systems. Several methods were attempted during the Edwards AFB demonstration but none provide an accurate measurement. The main problem is the surging action of the air lift pumping makes measurement difficult. The vendors are working to develop methods to measure flow rates in air lift systems, but until a technique is developed, only rough estimates will be available.

There are three types of data that have been used to determine the radius of the circulation cell (or location of stagnation points) including head measurements, contaminant data, and tracer migration. Unfortunately, there is often disagreement in the results from these data. For example, at the March AFB demonstration, contaminant concentration suggested that water was being circulated to 90 feet in all directions around the well while the tracer data indicated that there was only flow in the downgradient direction. Head measurements, measured as water levels in piezometers or monitoring wells around the circulation cell, are difficult to use because the head changes induced by GCW operation diminish rapidly with increasing distance from the well. It becomes difficult to measure the small head changes because of interference by natural groundwater fluctuations.

Decreased contaminant concentrations throughout the circulation cell is a good indication of circulation; however, the data rarely shows a evenly distributed and uniform pattern of decreasing contaminant concentrations. GCWs are very effective at smearing contaminant and the concentrations at monitoring locations around the site tend to fluctuate up and down. Superimposing these changes onto the background fluctuations in contaminant concentrations can make it difficult to verify circulation.

Tracer tests provide the best evidence for groundwater circulation. The bromide tracer test at Tyndall AFB showed that circulation was occurring which concurred with the trends shown in the contaminant profiles. The tracer tests at Hill AFB showed that the GCW was not circulating groundwater. Even though these are different outcomes, they are examples of successful applications of tracers for the intended purpose. The results from the tracer test at March AFB conflicted with contaminant concentrations. Which method was more accurate was not clear, but it would be expected that if the changes in contaminant concentration were due to communication with the GCW that the tracer would have shown up at all locations. This was not the case. What this does suggest is the need for a reliable tracer approach. Dr. Richard Johnson from the Oregon Graduate Research Institute developed the tracer test approach used at Hill AFB that is included in this protocol. The test includes dual tracers and both a convergent and divergent approach. The test will provide the data needed to verify and measure groundwater circulation.

Currently, the GCW technology is being applied to clean up sites and sites have achieved closure. However, the items discussed above need to be addressed to further develop the technology for effective and reliable use. At its current level of development, the technology is more appropriate for use in treating source zone contamination, or areas of more concentrated contamination, at sites with relatively simple hydrogeology (i.e., porous uniform aquifers with low background groundwater velocities) where the risk for exposure to sensitive receptors is minimal. Reliable use of the technology at sites with more complicated hydrogeology, or for use for plume capture an/or control, will require a more thorough site characterization than is currently conducted, improved modeling and design procedures, and improved system operating and performance monitoring. Currently, the risks of implementing the GCW technology for plume capture must be given strong consideration before selecting the technology for this purpose. Detailed pilot-scale tests must be conducted to determine if the required level of control and treatment can be achieved.

## 3.0 GCW Technology Screening Procedure

The following sections are provided to guide protocol users through a decision process to determine if implementation of the GCW technology is appropriate for their site. The material is based on information that was available at the time this protocol was prepared. The technology is continuing to develop with an increasing number of field applications, and as it matures, its potential may expand to include some applications not covered by this protocol.

### 3.1 Decision Tree and Process Description

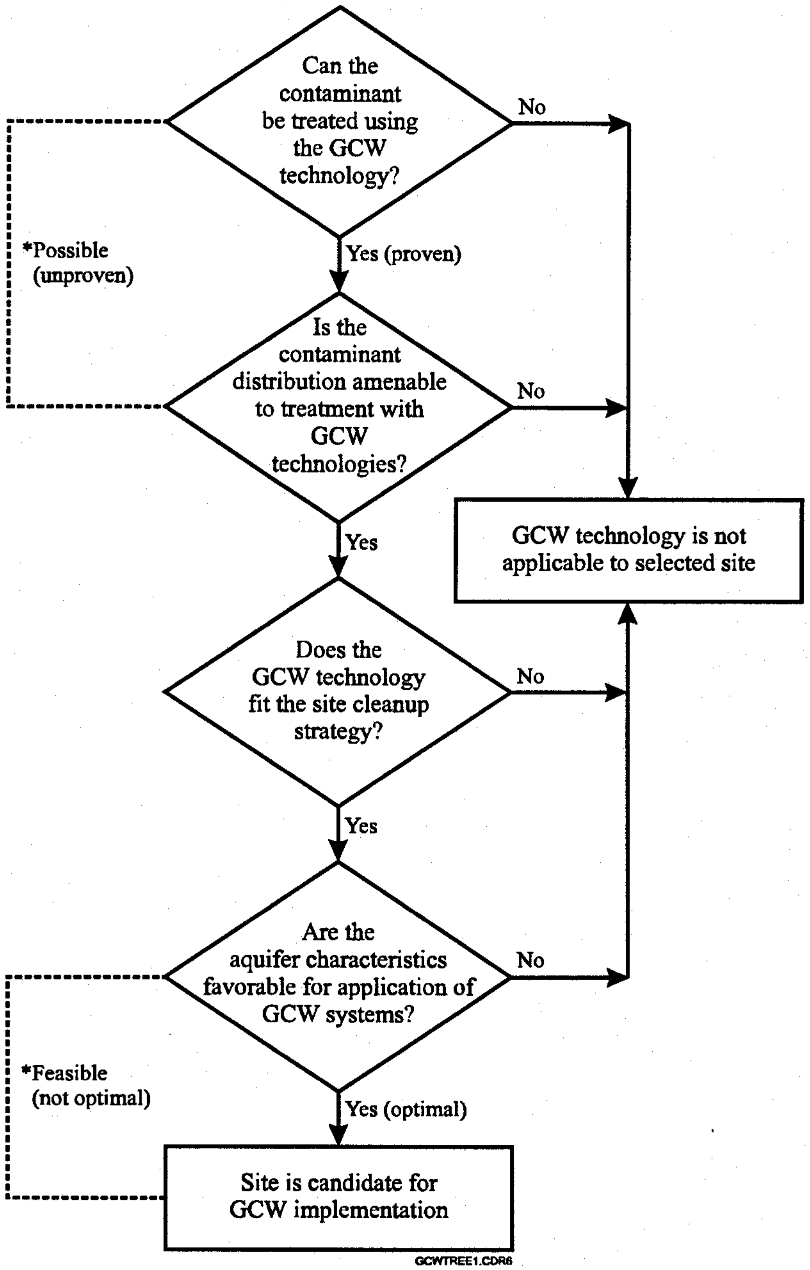
The process for screening the GCW technology is a logical sequence of steps during which site specific data is reviewed to make decisions as to whether or not the technology is applicable for that site. Figure 4 illustrates this process. The decision tree was formulated to assist in screening the technology based on information that is likely to be available or is easily obtained. As the decision process advances to the higher levels, the data requirements for making decisions become more complex and may require some additional site investigation work.

The decision to proceed with the technology can only be made after the process is completed. The decision not to proceed with the GCW technology can be made at any step when the criteria along the central path of the process are not satisfied. Remember that the outcome of this process is a decision to further consider GCW for a site. Although the process will provide an indication of the potential for a successful application, further site specific investigations such as are outlined in Volume II will be required to determine the actual potential of the technology at a site.

### 3.2 Decision Process

It is important for the protocol user to be aware of the technology's level of maturity. The decision approach described in the following sections contains information on the level of maturity of the technology for specific contaminants and cross references to case history information that is found in the Appendix.

Table 10 indicates the applicability of the GCW technology for different scenarios and the current status of the technology. The applicability ratings should only be used as a guide to determining the potential for screening purposes. The actual potential for success is dependent on many site-specific conditions and all of these must be taken into consideration before making any decision to proceed with any remedial technology. The status of the technology indicates the level at which the vendors have claimed successful application of GCW systems. It cannot be directly translated into the successful potential for any given site. The user of this protocol must carefully go through the decision process described below to determine if the GCW technology has potential for application at their specific site.




* Dashed lined indicate increased risk in treatment success.

**Figure 4. Schematic of the Decision Process for Screening Sites for GCW Applicability.**

**Table 10. Applicability and Status of the Groundwater Circulating Well Technology.**

| | Applicability | Status of Technology |
|---|---|---|
| **Contaminant Type** | | |
| Volatile organic compound (VOC) | ✓✓✓ | Proven |
| Semi-volatile organic compound (SVOC) | ✓✓ | Demonstrated |
| Metals | ✓ | Conceptual |
| Radionuclides | ✓ | Conceptual |
| **Clean-up Strategy** | | |
| Containment | ✖ | N/A |
| Source treatment | ✓✓✓ | Proven |
| Plume reduction | ✓✓ | Proven |
| Plume interception | ✓ | Conceptual |
| **Unsaturated Thickness** | | |
| 0-5 ft | ✓ | Demonstrated |
| 5-1,000 ft | ✓✓ | Proven |
| **Saturated Thickness** | | |
| 0 - 5 feet | ✓ | Demonstrated |
| 5 - 115 feet | ✓✓ | Proven |
| >115 feet | ✓ | Conceptual |
| **Aquifer Characteristics** | | |
| Porous media | ✓✓ | Proven |
| Fractured media | ✓ | Conceptual |
| Karst | ✓ | Proven |
| <u>Background Flow Velocity</u> | | |
| low (e.g., >0.001 ft/d) | ✓✓✓ | Proven |
| medium (e.g., 0.001-1 ft/d) | ✓✓ | Proven |
| high (e.g., >1 ft/d) | ✓ | Conceptual |
| <u>Horizontal Hydraulic Conductivity</u> | | |
| low (e.g., < 0.03 ft/d) | ✖ | N/A |
| moderate (e.g., 0.03 - 3.00 ft/d) | ✓✓ | Proven |
| high (e.g., >1ft/d) | ✓✓✓ | Proven |
| <u>Ratio of the Horizontal to Vertical Hydraulic Conductivity</u> | | |
| Isotropic (H:V <3) | ✖ | N/A |
| Anisotropic (H:V 3 - 10) | ✓✓ | Proven |
| Highly Anisotropic (H:V >10) | ✓ | Conceptual |
| <u>Aquifer chemistry</u> | | |
| High iron in water | ✓ | Proven |
| High calcium in water | ✓ | Proven |
| High magnesium in water | ✓ | Conceptual |
| <u>Soil chemistry</u> | | |
| High sodium absorption ratio (SAR) values | ✓ | Proven |

Key:

| | |
|---|---|
| Highest potential for success | ✓✓✓ |
| Good potential for success | ✓✓ |
| Potential for success based on special considerations | ✓ |
| Not applicable/Restricted Use | ✖ |
| Not applicable | N/A |

**3.2.1 Contaminant Type.** The first consideration in determining if GCW technology is suitable for remediating a site is whether or not the contaminant can be moved to the well for treatment or destroyed or degraded in the aquifer. Currently, the GCW technology is young. As the technology matures, it is likely that its applicability will broaden to include contaminants not on the list.

Section 3.0 describes a screening process that must be followed to determine if the GCW technology should be considered for any given site. Table 11 is a list of common environmental contaminants that are potential candidates for GCW treatment. The list was developed from the vendor provided information for their configurations (see Appendix). The table includes an indication of the maturity of the GCW technology for treating each contaminant, and provides cross-references to case histories found in the Appendix that deal with that contaminant. The information in the table should be reviewed with the understanding that these are generalizations based on current information and that the application of GCWs is contaminant-, site-, and clean up strategy-dependent. If the contaminant(s) from a given site are listed in Table 11, then the GCW technology may be applicable for that site and the decision process should progress to the next step.

**3.2.2 Contaminant Distribution.** Subsurface contamination is found in any of four phases: dissolved phase, sorbed phase, vapor phase, and free phase. Typically, GCW systems are more appropriate for dissolved-phase contaminants, which are more readily transported to the well. Once a contaminant is moved to the well, it can be removed from the aqueous phase through any of several treatment processes.

As the contaminants become more hydrophobic, or in aquifers characterized with high organic matter concentrations, sorption onto aquifer solids increases and contaminant transport to the well can be retarded. When this is the case, a GCW system design that provides a larger percentage of the remediation within the aquifer outside the well is appropriate. Most often, GCW systems that incorporate in situ treatment exploit biodegradation as the primary mechanism of contaminant reduction. These systems need to be designed to optimize the delivery of nutrients and/or electron donors or acceptors via the circulated groundwater.

Residual contamination is often present in the vadose zone at sites with contaminated groundwater. GCW systems are available that simultaneously remediate this residual in the vicinity of the well. This is accomplished by pulling a vacuum on the head of the well or by coupling SVE or bioventing to the GCW system. Soil vapor extraction volatilize sorbed-phase contaminants in the vadose zone and remove vapors for aboveground treatment. Bioventing enhances in situ biodegradation of vapor- and sorbed-phase contaminants and typically does not require aboveground vapor treatment. Both methods replenish soil-gas oxygen concentrations.

Free-phase contamination presents a challenge for GCW systems. GCW systems are not suitable for removing free-phase, light, non-aqueous-phase liquids (LNAPLs). This is because LNAPLs tend to float at the water table interface, and unless the GCW system can remove 100 percent of the free-phase liquid on its first pass through the well, the circulation system will smear the contaminant throughout the treatment volume. This can produce a more difficult remediation challenge. It is recommended that another technology, such as bioslurping, be used to remove the LNAPL first, then a GCW system can be employed to address any, residual dissolved- and/or sorbed-phase contaminant.

**Table 11. List of Contaminants Amenable to Treatment with Groundwater Circulating Well Systems along with Case History References.**

| Petroleum Hydrocarbons | | |
|---|---|---|
| **Contaminant** | **Potential** | **Case Histories (See Section III)** |
| Total Petroleum Hydrocarbon (TPH) | Strong | 1,6,24 |
| Benzene, Toluene, Ethylbenzene and Xylenes (BTEX) | Strong | 43,45 |
| Benzene | Strong | 2,3,4,5,6,7,8,9,10,11,12,13,14,15,16,17,19,22,27, 33,40 |
| Toluene | Strong | 2,3,4,8,11,12,13,16,17,19,27,33,40 |
| Ethylbenzene | Strong | 2,3,4,8,11,12,13,15,16,17,19,22,24,27,33,40 |
| Xylenes | Strong | 2,3,4,8,11,12,13,15,16,17,19,24,27,33,40 |
| **Non-Petroleum Hydrocarbons (PAHs)** | | |
| **Contaminant** | **Potential** | **Case Histories** |
| Methanol | Strong | 18, |
| Isopropyl Ether | Strong | 27,33 |
| **Polycyclic Aromatic Hydrocarbons** | | |
| **Contaminant** | **Potential** | **Case Histories** |
| Naphthalene | Strong | 2,3,4,7,8,11,13,14,15,16,17,33, |
| 2-ring PAH | Moderate | 44 |
| 3-ring or greater PAH | Weak | 44 |
| **Chlorinated Hydrocarbons** | | |
| **Contaminant** | **Potential** | **Case Histories** |
| Total Chlorinated Hydrocarbon | | 29,30,31,32,34,35,36,37,38,39,41,42 |
| Dichloroethene | Strong | 1,22 |
| 1,1-dichloroethane | Strong | 28 |
| 1,2-dichloroethane | Strong | 28 |
| 1,1,1-trichloroethane | Strong | 28 |
| | Strong | 25,26, |
| Perchloroethylene | Strong | 28 |
| Trichloroethylene | Strong | 21,23,28 |

Free-phase, dense, nonaqueous-phase liquids (DNAPLs) are different than LNAPLs in that they do not float at the water-table interface, but tend to sink, forming ganglia or small pools of free-phase DNAPL within the aquifer. Formulating cleanup strategies for free-phase DNAPL is very difficult because of its irregular distribution. Regardless of the selected technology, methods that utilize groundwater movement to entrain and remove the free-phase DNAPL will result in some degree of smearing, and their successful application will depend on the solubility and mass transfer properties of the contaminant in the aquifer. Because of this, and depending on site-specific conditions, GCW systems may be among the more effective methods for treating free-phase DNAPL contamination.

**3.2.3 Clean-up Strategy.** It is important to develop a comprehensive clean-up strategy before choosing any remediation technology. The primary objective should focus on the protection of human health and the environment. Also fundamental to an effective cleanup is the removal of the hotspot or "source" of the contamination in the soil and/or groundwater near where the contaminant entered the ground. This is known as source term reduction. The residual contamination acts as a source that continues to disperse in the environment if left untreated. Removal of the source often provides the most protection to the environment and is the most effective use of clean-up dollars. Other strategies include plume interception, hydraulic containment, wellhead treatment, intrinsic remediation, or no-action. The particular clean-up strategy employed is site- and situation-specific and is generally based on protection of human health and the environment, effectiveness, and cost.

**3.2.3.1 Contaminant Source Treatment.** The characteristics of GCW systems make them best suited for source term reduction because of their 1) effectiveness in cleaning up relatively high concentrations of contaminant, 2) "flushing" created by inducing a vertical gradient across layers of lower permeability, thereby reducing residual contamination, and 3) relatively limited zone of influence.

Vapor stripping is a physical process that transfers dissolved VOCs from the aqueous phase to the vapor phase and is dependent on the initial aqueous concentration of the contaminant. The vapor stripping capacity or effectiveness can be assessed by measuring the percentage of contaminant converted into the vapor phase per pass through the well.

A primary advantage of GCW systems are that the circulation cell created in the subsurface establishes a vertical gradient that directs flow vertically across lower permeability zones that often contain a significant percentage of the contamination. In pump-and-treat systems, water is preferentially pulled in horizontally from the higher permeability zones and the contamination in the lower permeability layers is not substantially addressed. By not affecting the contamination in the lower permeability zones, the contaminant continues to diffuse out of these sediments over time, creating a diffusion-limited problem that can result in long cleanup times. Although not fully proven, it is likely that if flow flushes vertically through these lower permeability zones, the clean-up time could be significantly reduced.

**3.2.3.2 Plume Reduction.** GCW systems have been used successfully to remove contamination from dissolved plumes. In situations where the background groundwater velocities are slow, the systems must be designed so that the entire extent of the plume is covered by the GCW-induced circulation cells. At sites where the groundwater is moving slowly, GCW placement can exploit this movement to facilitate contaminant transport to the well.

**3.2.3.3 Plume Interception.** Recently GCW applications have expanded to include plume interception or capture. In plume capture scenarios, wells are placed in a line perpendicular to the path of the plume to both intercept and contain the movement of contamination. Depending on the width of the plume to be captured, and the level of treatment required, a relatively large number of GCWs may be needed to adequately capture the plume and the relative cost-effectiveness may be decreased. It should be noted that hydraulic control is much more difficult to predict and manage with GCW systems versus conventional pump and treat. Because of this, there is some inherent risks associated with using GCWs for plume capture or interception.

**3.2.3.4 Hydraulic Containment.** If hydraulic containment is required, a standard pump-and-treat system is generally more effective. A GCW will not provide good containment because a hydraulic "sink" is not created. All the water pumped from one zone is reinjected into an adjacent zone at the same location. There is no net extraction at the well for hydraulic containment. In practice, the system relies on the natural regional gradient to bring new contaminated water into the circulation cell and carry clean water downgradient.

**3.2.4 Hydrogeologic Considerations.** The most critical factor that influences the operation of a GCW system is the geological setting in which it is installed. The circulation cell that is driven by the GCW system is an in situ process, and the surrounding environment has the greatest influence over its development and flow characteristics. Therefore, to effectively design, install, and operate any GCW system, an adequate evaluation of the hydrogeologic environment must be conducted.

**3.2.4.1 Vadose Zone (Unsaturated) Thickness.** The thickness of the vadose zone has a direct impact on the selection and applicability of GCW systems. By definition, the unsaturated thickness is the distance between the ground surface and the water table. It can be determined by measuring the depth to groundwater in a monitoring well that is screened across the water table surface. The thickness of the vadose zone must be sufficient to a) permit the recharge of the circulated groundwater and b) to provide sufficient vapor residence times for systems that discharge the system off-gas for treatment in the vadose zone. The required thickness will vary depending on the specific GCW configuration, the permeability of the treatment zone, the groundwater pumping rate, and the air flowrate.

In general, the vadose zone should be at least 10 feet thick for most GCW applications; however, GCWs can be applied at sites with a thinner vadose zone provided that circulated groundwater can be recharged without mounding to the surface. There are system modifications that can be incorporated into certain GCW designs to allow application in shallow vadose zone settings. For example, an infiltration gallery or trench may be installed near the surface at sites with a shallow (e.g., 5-10 ft) vadose zone. These sites may not be amenable to vadose zone discharge of the vapor due to unacceptable emissions of vapor-phase contaminant from the ground surface to the atmosphere.

For GCW configurations that employ an infiltration gallery to reintroduce the treated water back into the aquifer, the achievable infiltration rate of the circulated groundwater through the unsaturated zone above the water table may be the governing factor controlling the pumping rate of the system. Because the pumping rate has a direct effect on the volume or zone of influence, the rate must be sufficient for the GCW system to be effective. The capacity of the unsaturated zone is a critical design parameter for GCW systems and must be calculated and field tested to determine a design groundwater-pumping rate.

GCW systems that discharge their off-gas into the vadose zone require sufficient vadose zone thickness to provide the residence time required for biodegradation of the vapor phase contaminant. The thickness required will be dependent on the porosity, permeability, and biodegradative capacity of the soils within the zone of treatment. Field tests described in Volume II can be conducted to determine each of these parameters.

Sites characterized with a 10- to 30-foot-thick vadose zone are better candidates for GCW implementation. These sites have sufficient thickness to allow for water circulation and for effective design of vapor movement and/or SVE components of the GCW system. Systems that incorporate bioventing could be applicable at sites with 10- to 30-foot-thick vadose zones provided the vapor residence times in the vadose zone are sufficient to allow the necessary degree of biodegradation to occur. If residence times are too short, discharge limits may be exceeded and off-gas treatment may be required.

A 30-foot or greater vadose zone thickness is optimum for most GCW applications. This thickness allows for optimum circulation of groundwater as well as optimal design and operation of auxiliary systems, including either SVE or bioventing.

**3.2.4.2 Saturated Thickness.** The saturated zone thickness is the distance between the water-table surface and the bottom of an aquifer. It is usually determined during well installation or other drilling activities when the depth to the bottom of the aquifer can be measured. The saturated thickness must be sufficient to accommodate the physical dimensions of the GCW. In air stripping GCW systems, the contact time of the air with the water must be maximized to facilitate the most effective stripping system. In some well designs, this requires the casing length to be long enough to allow volatile compounds to partition into the vapor phase. The depth to the aquifer is not as critical as the saturated thickness as far as the groundwater circulation is concerned. However, compared to a pump-and treat system the relative energy savings of a vapor stripping system may actually increase with depth, because with a GCW, the water need not be lifted to the surface for treatment.

**3.2.4.3 Aquifer Characteristics.** GCW systems are most effective for treatment of contamination in porous media, such as sands and gravels. In some cases, fractured rock can be treated if there is sufficient fracturing and the fracture sets are oriented to facilitate re-circulation. Karst aquifers can also be treated. The effectiveness of fracture and karst treatment is a site-specific constraint.

**3.2.4.4 Hydraulic Conductivities.** The most important site geologic characteristics that will affect the operation of the GCW are the horizontal and vertical hydraulic conductivities. In general, GCW installations are most effective at sites with horizontal hydraulic conductivities greater than $10^{-3}$ cm/sec. Sites characterized with horizontal conductivities between $10^{-3}$ cm/sec and $10^{-5}$ cm/sec are potential candidates and the decision process should proceed to the next step. For unconsolidated porous media, this range of conductivity would comprise silty sands or coarser sands and gravel. Consolidated media include sandstones, limestones, and fractured igneous and metamorphic rocks. Selection or rejection of the GCW technology at these sites may depend on other site characteristics. Remember that lower hydraulic conductivity will result in a smaller ROI and may require more wells. Application at sites with hydraulic conductivities less than $10^{-5}$ cm/sec is not recommended.

The ratio of the horizontal to vertical hydraulic conductivities ($K_H$:$K_V$) is a measure of the anisotropy of the site. Typically, the vertical hydraulic conductivity is lower than the horizontal hydraulic conductivity in stratified or layered formations. The primary cause of anisotropy on a small scale is the orientation of clay minerals in sediments and unconsolidated rocks (Freeze and Cherry, 1979). For the optimum application of a GCW system, an overall ratio of the horizontal to vertical hydraulic conductivity, or anisotropy, should be in the range of 3 to 10. Sites with lower ratios may be candidate sites but the dimensions of the circulation cell will tend to round and the ROI will approach the distance between the intake and reinjection screens. Sites characterized with ratios greater than 10 could run a risk of not circulating the water due to the increased resistance to vertical flow. The net result could be horizontal flow to the intake screen and horizontal flow from the reinjection screen. Sites with rations approaching 100 or greater be excluded from consideration.

**3.2.4.5 Stratigraphy.** Layering is beneficial in most cases because less permeable layers promote horizontal flow, thereby increasing the radial influence of the system. In the extreme cases, however, impermeable layers between the extraction and injection screens may prevent circulation back to the extraction screen resulting in "infinite" horizontal flow of the re-injected water. These impermeable layers may be very thin (<6") and for this reason it is suggested that at least one characterization well be cored and examined in detail to identify any zones that may prevent circulation. In the other extreme, if the anisotropy is less than 3, (the horizontal and vertical hydraulic conductivity are nearly the same) the volume of the circulation cell will be greatly decreased. The re-injected water will flow toward the extraction screen with very little lateral movement, rendering the technology inefficient due to the number of wells that would be required to cover the contaminated aquifer.

**3.2.4.6 Background Groundwater Velocity.** Background groundwater velocity refers to the natural speed that the groundwater is moving through an aquifer past a GCW (see Section II-2.2.1.6). Depending on the application, GCW systems can be employed over a range of background groundwater velocities.

At sites where groundwater is stagnant or moving very slowly, GCWs may prove effective for source reduction. Under this scenario, the system is designed so that the total zone of influence covers the zone requiring treatment. The circulation of the groundwater is the only mechanism for transporting the contaminant to the well, or for delivering oxygen, nutrients, or other compounds to the aquifer.

The maximum background flow velocity of the groundwater needs to be in the range of the system's ability to capture the contamination, and in most cases to allow multiple passes in the well before it moves downgradient. The ability of the GCW to capture the groundwater will be related to the pumping rate of the well. The minimum background flow velocity needs to be sufficient to prevent stagnation around the well. Enough water needs to be moving past and through the system to carry the clean water downgradient.

**3.2.5 Decision Summary.** The decision to proceed with GCW implementation can proceed once all of the items discussed in the preceding sections have been considered. If all of the criteria were met and there no concerns raised, the technology should be considered. If for any reason the technology did not meet any of the described criteria, it should be dropped from consideration and an alternative technology should be pursued. The risks associated with proceeding with the technology will be too high and the potential for a failed application too great. GCW systems are much more complex than conventional pump and treat systems, and although GCWs offer several distinct advantages, their installation and even pilot-scale testing can be expensive so there must be a significant level of confidence that the technology will work at a given site.

# Volume II

# Technology Implementation Guidance

## 4.0 Groundwater Circulating Well System Design Requirements

The design approach consists of a stepwise iteration in which more detail is added to the design as more information is collected. The approach is summarized in the outline below:

**I. Preliminary Design**

**A. Conceptual model development:** Assembly of all pertinent site- and contaminant-specific data into a coherent foundation on which to build design

**B. Design estimates:** Preliminary design calculations to estimate required pumping rates, number of wells, remediation time, aquifer impacts

**C. Model selection and construction:** Fitting the type of information required and produced by a numerical model to the design information required, and the site information available

**II. Pilot-Scale Testing:** Installation and field testing a GCW

**III. Full-Scale Design:**

**A. Model calibration and execution:** Using field data to adjust model inputs and predict GCW performance

**B. Well placement:** Using model and field data to determine appropriate GCW locations

**C. System component selection:** Selecting parts for the design that achieve design goals

**D. Monitoring network:** Selecting the type and location of monitoring devices in the GCW and the soil formation.

This design process is dependent on engineering and cost constraints. For example, funding resources for a site needing only one GCW unit to effect treatment may preclude the use of both modeling and pilot-scale testing. The method that will produce the most design data should be selected.

The effectiveness of the GCW technology is dependent on the ability to move and circulate water in a natural flow system. The circulation pattern established around a GCW is dependent on the characteristics of the geologic materials, the characteristics of the ambient flow field, the design of the GCW system, and the hydraulic stresses that are imposed on the natural flow field. Because of this, it is essential to obtain and review site-specific geologic and hydrogeologic information. In addition, the operation of a GCW system will impart a complex hydraulic response in the aquifer that is best understood through monitoring and groundwater flow modeling. It is critical to the successful application of a GCW system that the aquifer characteristics and hydraulics are well characterized, monitored, and evaluated.

The objectives of the site characterization, system design and groundwater flow modeling work associated with the installation of a GCW are as follows:

- to select, design, and apply a GCW system that is compatible with the site geology, hydrogeology, and contaminants of concern
- to optimize the system design and pumping rates to maximize the hydraulic zone of influence associated with the GCW installation
- to properly install and operate the system based on the site data, modeling results, and remediation needs.

An approach to achieving these objectives can be summarized by the following steps:

- review the available site characterization data and fill in any gaps with information obtained from coring and analysis, aquifer testing, and determining site background conditions
- perform a pilot-scale field test and conduct aquifer testing
- select a general GCW configuration based on the site geology, hydrogeology, and type and distribution of contaminant(s)
- construct and apply a groundwater flow model to evaluate the proposed GCW design and operation
- implement any necessary changes in the configuration/design based on the field test results and flow modeling
- install the GCW system and monitoring equipment
- perform a system shakedown to ensure proper operation.

### 4.1 Required Site Data

Typically, some level of site characterization data exists for any site for which a GCW application is being considered. However, intricate site-specific data is required to determine the suitability of the GCW technology, and for designing a system for that site. Most often, this data is not available. For this reason, it is necessary to conduct a thorough site investigation to collect the data needed to screen the technology. This includes installation of a minimum of 3 multi-level, multi-purpose wells. More of these wells will be required at sites requiring multiple GCW installations. The wells are designed for use as both characterization and system monitoring wells.

**4.1.1 Site Characterization Requirements.** The following is the approach and rationale for conducting a site characterization for GCW implementation. First, core a well at the proposed remediation location. Evaluate the core for any sedimentary layers that could restrict circulation. Sections of the core are then selected for laboratory analysis to obtain both hydraulic and chemical data required for the system design. Following the core analysis, a suite of aquifer tests must be conducted to obtain the following parameters: horizontal and vertical hydraulic conductivity, and sustainable pumping and injection rates. Aquifer testing should include a circulating test (dipole test) that is conducted in a single well to simulate GCW system operation. The decision to continue with the design of the GCW system or consider alternative treatment options is based on the site-specific data. If the decision is made to continue, the characterization wells are then monitored to establish background conditions at the site, and the wells are used with the full-scale system to monitor system performance during operation.

**4.1.2 Soil Core Requirements and Analysis.** Characterization data from the wells must include a detailed examination of the sediment types and stratigraphy. Low permeability layers must be particularly noted and tested in the laboratory. In some cases, even very thin (e.g., <6") layers can be sufficient to prevent circulation or significantly alter the flow paths. Subcores selected for testing must be representative of each layer, and laboratory testing must be designed to obtain the hydraulic conductivity of each layer. Laboratory testing may include falling head tests (Klute, 1965), unsaturated flow analysis (UFA) (Wright and Conca, 1994), or grain-size analysis to determine hydraulic conductivity (Koltermann and Gorelick, 1995).

In addition to the hydraulic testing, chemical analysis must be performed to determine the concentration and distribution of the contaminants and general groundwater and soil chemistry. Samples are collected across the anticipated circulation zone and include both sediment and water samples. The sediment samples can be collected from the core, and water samples can be collected during and/or after drilling. The samples must be analyzed for contaminants and general chemistry (e.g., anions, cations, cation exchange capacity, and total organic carbon). It is important to know the pre-treatment concentrations and distributions in order to interpret how the operation of the treatment system affects the subsurface environment. For example, the chemistry results could provide information on potential plugging problems that could result from high concentrations of carbonates or metals (e.g., iron) in the water.

**4.1.3 Aquifer Testing Requirements and Procedures.** A series of aquifer tests must be conducted to provide at least the following parameters: horizontal hydraulic conductivity, vertical hydraulic conductivity, and sustainable pumping and injection rates. Good testing and analyses are critical to the design process. Results will provide the most important parameters (horizontal and vertical conductivity) used to design the GCW system. A relatively small investment in characterization up front can save significant remediation dollars later.

**4.1.3.1 Pumping Tests.** Single-well pumping tests involve pumping water from a test well and measuring both the discharge from and the drawdown in the well over time. In multiple-well pumping tests, the drawdown in other monitoring wells and/or piezometers at known distances from the pumping well can also be monitored and included in the test. Discharge and drawdown measurements are then substituted into appropriate well-flow equations to determine the aquifer hydraulic conductivity.

Well-hydraulics equations that model the response of specific aquifer types (confined, leaky confined, or unconfined), can be used to describe the flow characteristics of the aquifer. The well-flow equation used must match the aquifer type found at the contaminated site at which the GCW remediation system is proposed. It is also important to note the assumptions and boundary conditions upon which the well-flow equation is based. These assumptions may include the following:

- aquifer type
- penetration depth of the pumping test well into the aquifer (most equations assume full penetration to the bottom of the aquifer)
- homogeneous aquifer material
- isotropic aquifer
- well-bore storage volume (usually assumed to be negligible)
- presence of recharge or impermeable boundaries.

Information gathered during the site characterization will be needed to properly interpret the results of the pumping test. Subsurface geological features, the aquifer type, the three-dimensional extent of the aquifer (especially confining and recharge boundaries), historical water-table levels, the regional hydraulic gradient, and the existence of nearby production or injection wells are all topics that must be reviewed in the site characterization.

Information regarding the well construction is also essential to the test. The well diameter, while it is important that it be defined, is not a critical parameter of the well construction. Well yield only increases by about 10% with a doubling of the well diameter. For newly installed wells, the well diameter should be designed to accommodate the pump planned for use during the pumping test. The screen slots themselves should not cover over 40 percent of the well circumference. This ensures that water enters the well with a low velocity and makes frictional head losses minimal. When feasible, the screened section of the well used for the pumping test and the analytical methods used to interpret results should be compatible with the GCW design intended for the treatment system. For example, if the proposed GCW design penetrates 20% of the total aquifer thickness, a well of similar dimensions should be used to pump water, and a well-hydraulics equation that incorporates the assumption of a partially-penetrating well should be used to analyze the test data.

The pumping test involves pumping water from a test well while carefully monitoring both the discharge rate and the water level in the test well. Pressure transducers piezometers can facilitate frequent and reliable water level readings. The quantity and quality of information provided by the test can be greatly increased if nearby observation wells, monitoring wells, or piezometers can be utilized during the test. If nearby wells are available, the water level in these wells must be recorded during the test.

The frequency with which discharge rate and water level data are measured and recorded decreases as the test progresses. Initially, water level measurement are made every 10 seconds. After 2 minutes, measurements can be taken every 30 seconds, and so on. After five hours of pumping, water level measurements only need to be recorded every hour. While the discharge rate is being measured and recorded, the goal is to keep the discharge rate constant throughout the duration of the pumping test. Pumping tests are conducted for 72 hours in unconfined aquifers. Shorter tests may be applicable when testing in a confined or leaky confined aquifer.

The interpretation of pumping test results is typically accomplished by graphical analysis of the drawdown versus time plots of the test data. The data is typically plotted as log drawdown versus log time, and curve-fitting techniques are commonly used to characterize the data. Interpretation of the characteristic curves fit to the data varies by method, but usually involves the comparison of data curves to template curves that represent various aquifer parameter values. The information typically produced by these analysis methods includes aquifer hydraulic conductivity, transmissivity (product of hydraulic conductivity and saturated thickness), and storativity. Some methods yield both the horizontal and vertical hydraulic conductivity of the aquifer. A complete description of the theory and application of pumping tests can be found in Domenico and Schwartz (1990) and Fetter (1994). A complete description of pumping tests and the various methods that can be used in the analysis of data collected during a pumping test is provided in Kruseman and de Ridder (1991). A guide is available to aid in the selection of the proper aquifer test techniques for a given aquifer type (ASTM D 4043-91).

While pumping tests generally give reliable information on hydraulic conductivity, they may be difficult to conduct in contaminated areas because the water produced during the test generally must be contained and treated as investigation-derived waste (IDW). In addition, a minimum 4-inch-diameter well is generally required to conduct pumping tests in highly transmissive aquifers because the 2-inch submersible pumps currently available are not capable of producing flow rates adequate to induce significant drawdown. In areas with fairly uniform, or homogeneous, aquifer materials, pumping tests may be conducted in uncontaminated areas, and the results can be used to estimate hydraulic conductivity in the contaminated area.

It is important that pumping tests be conducted in wells that are capable of yielding the volume of water being pumped during the tests and do not constrict flow into the well. If this occurs, the pumping test is not testing the ability of the aquifer to transmit water to the well, but rather the transmissive capacity of the well screen in the test well.

**4.1.3.2 Dipole Testing.** The dipole test was developed to provide information on the local vertical distribution of horizontal and vertical hydraulic conductivities and the vertical distribution of the specific storativity (Kabala, 1993). Although the test was developed for confined or leaky confined aquifers, the small amount of drawdown makes it appropriate for unconfined aquifers as well.

For GCW design, the test is superior to conventional pump test methods because groundwater in circulated in the formation much like during GCW operation. Another advantage of the dipole test is that water is not withdrawn from the ground, eliminating disposal requirements. It is strongly recommended that dipole testing be conducted when designing GCW systems because the data gathered is essential for proper design, and the test simulates GCW operating conditions. The following is a brief discussion of the basics of the dipole test, a more detailed description of the test and data analysis procedures can be found in Kabala 1993.

The dipole test is conducted in a single well. At sites where multiple wells are required, or where the ROI is expected to be large, tests in multiple wells may be necessary. The tests include isolating a vertical section with two inflatable packers. The isolated section is divided into two chambers with an third inflatable packer modified to allow water to be pumped from one chamber to the other. It is critical that this packer is installed properly to provide a tight seal between the two chambers. A bentonite seal is required in the annulus of the well adjacent to the middle packet to prevent interference from the "skin" effect. A submersible pump is placed in one of the chambers. Pressure transducers are placed in each chamber to measure the pressure developed during pumping and above and below the two outside packers to monitor for leaks.

During the test, groundwater is pumped between the two chambers at a constant rate. This causes groundwater to enter the first chamber, then leave the well through the second chamber. The net result is circulation of the water in the formation just outside the well. During pumping, pressure readings from the four pressure transducers are recorded. Data loggers are used to rapidly collect the data through the transient and up to steady state conditions.

The data from the test is used to calculate the horizontal and vertical hydraulic conductivities, and the storativity. The pressure (drawdown) data are graphed against time. Three points are selected from different phases of the drawdown curve to define a system of three nonlinear equations. The three equations are solved through application of the Newton-Raphson algorithm for three unknowns. If any of the parameters are known from previous aquifer tests, a simpler computational method based on the Newton-Raphson iterative algorithm for one or two unknowns can be applied to find the remaining parameters.

**4.1.3.3 Slug Tests.** Slug withdrawal or injection tests are commonly used as alternatives to pumping tests and determine the hydraulic conductivity of an aquifer in the region around a well. Advantages of slug tests include the relatively short duration of the test, lasting only minutes to at most a few hours. In addition, no pumping is required, minimizing the volume of IDW generated by the aquifer investigation. Wells operating near a site where a pump test would normally be performed can interfere with pump test results. However, since slug tests indicate aquifer conditions in the immediate vicinity of the test well, the interference of nearby operating wells is also minimized.

The fact that slug test results reflect only the conditions in the aquifer in the immediate vicinity of the test well may be a disadvantage in heterogeneous aquifers. The information related by a slug test may not be representative of the general aquifer conditions. Pump tests involve a larger volume of aquifer, and therefore they provide a more representative picture of regional aquifer conditions. In addition, the soils immediately surrounding a well may have been disturbed during drilling, and therefore not be very representative of the undisturbed aquifer sediments.

A slug test is performed by adding or removing a "slug" of known volume to (or from) a well and monitoring the water level in the well as it falls (or rises) back to the equilibrium water level. The slug may consist either of water or a solid object (usually a cylinder) of known volume. If a water slug is removed, the test is sometimes referred to as a "bail test." In any case, the displaced water imposes a temporary hydraulic gradient around the well, which motivates water flow in the direction that will result in the restoration of the equilibrium water-table level.

As with pumping tests, slug test data must be interpreted using appropriate equations that take into consideration reasonable assumptions and boundary conditions. The assumptions and conditions must be matched as closely as possible to the actual aquifer, well, and test conditions during the slug test. Because slug tests are, in effect, sampling a small portion of the aquifer, more restrictions are imposed on the equations that describe the well response. Conditions that must be matched among the well, aquifer, and equation include the following:

- aquifer type (confined, unconfined, leaky confined)
- aquifer areal extent (commonly assumed to be infinite)
- aquifer heterogeneity (common assumption: aquifer is homogeneous)
- aquifer thickness uniformity
- aquifer isotropicity
- head in well changed instantaneously at $t_0=0$
- well penetration into aquifer (complete or partial)
- state of flow to (or from) the well (steady or unsteady state)
- wall and inertia effects in the well
- hydraulic gradient in test well region
- well storage (may not be neglected as in pumping test)

An alternative type of slug test is the oscillation test in which an imposed pressure is used to lower the water level in the well. The pressure is rapidly released and the water level in the well is carefully monitored as it rises and falls in a damped oscillation. The water movement in this test is more rapid and should be monitored with automated devices to ensure good data capture. The oscillation test does not neglect the inertial forces affecting the water level in the well, but includes many of the other assumptions and conditions of the other slug tests (Kruseman and de Ridder, 1991).

Slug tests may be used as a preliminary test to estimate aquifer conditions, or as a replacement to pumping tests if the volume of IDW or the presence of interfering wells is a problem. Either pumping tests or slug tests can be used to estimate the contaminated aquifer's hydraulic conductivity, which is an important parameter in GCW design and application.

**4.1.4 Background Monitoring Requirements.** Following testing and prior to startup of the GCW system, background conditions for the site must be determined by continued monitoring of the groundwater chemistry and hydraulic head in the site characterization wells. These background data will be important for interpreting the changes in groundwater chemistry and hydraulics during the operation of the GCW system and for interpreting the system's effectiveness in removing contamination from the subsurface. The characterization wells should then be used as monitoring locations to measure pressure heads (water levels) and sample for groundwater chemistry during and after the operation of the treatment system.

**4.2 System Design Requirements**

GCW design must be based on the site-specific geologic and hydrogeologic conditions, on the results of the field testing, and on the well configuration selected. Design parameters that must be taken into consideration include the following:

- general GCW configuration
- operable pumping rates
- well screen lengths and placement
- well diameter (based on the necessary diameter required for specific pump size, eductor pipe, and/or in-well treatment unit), and
- treatment approach.

Detailed GCW design is performed by the vendor of the specific configuration chosen. Design and installation of an independent GCW system must take patent restrictions into consideration.

**4.2.1 Preliminary Design.** The first step in implementing any remedial system is to develop a preliminary design. The following sections describe the process for developing a preliminary design for a GCW system.

**4.2.1.1 Conceptual Model Development.** The first step in any modeling effort is the development of a conceptual model. The conceptual model is a three-dimensional representation of the groundwater flow and transport system based on all available geologic, hydrogeologic, and geochemical data for a specific site. A complete conceptual model will include geologic and topographic maps of the site, cross sections depicting the site geology/hydrogeology, a description of the physical and chemical parameters associated with the aquifer(s), and contaminant concentration and distribution maps. The purpose of the conceptual model is the integration of the available data into a coherent representation of the flow system to be modeled.

The conceptual model is used to aid in model selection, model construction, and interpretation of model results.

**4.2.1.2 Design Estimates.** The required number of wells and pumping rates depends on the objectives of the GCW installation, the effective zone of influence for each well, and the distribution of the contamination requiring treatment. For source treatment and/or plume reduction, it is important that a sufficient number of wells be included in the design of the GCW system to cover the zone of contamination. If the objective of the GCW system is to intercept a migrating plume, a sufficient number of wells will be needed to effectively capture and treat the entire width, and potentially the length, of the contaminant plume as it passes through the GCW system.

Several analytical and semi-analytical approaches are available for estimating the number of wells and pumping rates for a GCW application. One such approach is the dimensionless capture area and flux method described by Philip and Walter (1992). These approaches are sufficient for preliminary design and for cost estimating purposes but must be reinforced with an evaluation of the system design using a numerical flow model. Numerical flow model and particle tracking techniques are used to ensure that the zones of influence overlap slightly, and to ensure that the objectives of the GCW system are achieved. Numerical models provide the most complete analysis and design of a GCW system and allow for investigations into different system configurations.

**4.2.1.3 GCW Configuration Selection.** There are numerous GCW configurations commercially available (see Appendix). When selecting a GCW configuration for a specific remediation effort, the type and distribution of the contaminant to be treated, as well as the hydrogeologic properties of the aquifer and vadose zone must be considered. Selection of the proper configuration is important to prevent smearing and redistribution of the contaminant(s). The vendors listed in Section III should be contacted to inquire about their specific GCW configurations and applications.

Selection of a GCW configuration depends on the type and distribution of contaminant to be treated, and the hydrogeologic properties of the aquifer and vadose zone. Use of the an inappropriate GCW configuration could cause smearing and redistribution of the contaminant, resulting in a contamination problem that is much worse than the original problem and dramatically increasing clean up costs.

Contaminant type and distribution dictates the appropriate operational mode for the GCW. Typically, dissolved LNAPL contamination is distributed toward the top of an aquifer. If the thickness of the contamination is such that the lower screen of the GCW extends below the contamination, a downflow mode is required to prevent injecting contaminants into clean zones of the aquifer. To meet regulatory requirements, the GCW may need to effect one pass treatment. Operating the GCW in a downflow mode draws the LNAPL contaminant into the upper section of the well. The water is treated in the well, then discharged to the aquifer through the lower section. If the lower screen lies within the contamination, either an upflow or downflow mode may be acceptable. As previously mentioned, use of GCWs is not appropriate when LNAPL free product is present. It is desirable to remove the product using a technology such as bioslurping prior to implementation of GCWs.

Conversely, DNAPL contamination tends to sink in an aquifer, resulting in dissolved DNAPL, unevenly distributed ganglia, and/or pools on low permeable strata. Most often, GCW systems are operated in an upflow mode when treating DNAPL contamination. It is highly unusual to operate a GCW at a DNAPL site in a downflow mode because of the risk of smearing and/or spreading the contaminant.

The properties of the aquifer, including the horizontal and vertical hydraulic conductivity, the stratigraphy of the aquifer materials, and the sustainable pumping/recirculation rates, are important factors that will govern the flow field resulting from the operation of the GCW system. To obtain the necessary flow rates and velocities needed to assure circulation, the well must be configured with proper well screens and have a diameter large enough to enable the placement of an adequate pump or drop tube for air lift pumping.

The permeability, thickness, and biological properties of the vadose zone are important characteristics that affect the selection of the appropriate GCW configuration. Systems that utilize the vadose zone for treatment of their off-gas require a sufficiently long vapor residence time in the vadose zone to achieve the desired level of contaminant destruction. The rate of biodegradation in the vadose zone must exceed, or at least equal, the mass input from off-gas in order to avoid exceeding atmospheric emission regulations. Systems that bring off-gas vapors aboveground for treatment can be applied at sites independent of the vadose zone thickness or biodegradation capacity. Specific well configurations are available from the various vendors.

**4.2.1.4 Pumping Methods and Rates.** GCW systems utilize two different methods for moving groundwater, mechanical pumping and air-lift pumping. The operating mode (upflow or downflow) is dependent on the water pumping mechanism which is often configuration specific. Mechanical pumping can be incorporated into GCW systems and enable the operation of the GCW to be in either an upflow or downflow mode. Most air lift systems operate in an upflow mode.

The pumping rate achievable by any GCW system is dependent on the site. The rate at which water is pumped by the GCW will have a direct impact on the hydraulic residence time in the well, the dimensions of the circulation cell, and the local groundwater velocities. Pumping rates must be optimized to meet the system's treatment objectives.

**4.2.1.5 Well Screen Selection and Placement.** Well screens should be selected to accommodate adequate volumetric flow rates. It is also important to select screens made of a material that is compatible with the contaminant(s) and aquifer geochemical characteristics. GCW vendors should be responsible for selecting the proper well screens.

The placement of the upper and lower screens in a GCW are determined based on the following:

- the GCW configuration selected and the type of circulation needed (i.e., circulation through the vadose zone or circulation within the saturated zone)
- the distribution and depth of contamination
- the mode of operation (i.e., upflow, standard circulation mode or downflow, reverse circulation mode)
- characteristics of the aquifer, including the stratigraphy, horizontal hydraulic conductivities, vertical hydraulic conductivities, and anisotropy of the aquifer materials.

**4.2.1.6 Circulation Cell.** The circulation cell is the volume of aquifer within the effective treatment radius of the GCW. The effective treatment radius is the furthest distance at which groundwater is circulated within an acceptable time period as defined by project goals. The circulation cell can be determined through field measurements of hydraulic head, the application of in situ flow meters, and the application of a three-dimensional groundwater flow model that incorporates the site geology, hydrogeology, and design parameters associated with the GCW system.

**4.2.1.7 Model Selection and Construction.** Mathematical models can be used in the design process of a GCW. Mathematical models simulate groundwater flow indirectly by means of a governing flow equation that represents the physical processes that occur in a natural system, along with equations that represent hydraulic heads or flows along the boundaries of the model area (Anderson and Woessner, 1990).

Mathematical groundwater modeling is a necessary step in the design and evaluation of large (multi-well) GCW systems because of the complex nature of the flow field that results from their operation. As a predictive tool, a groundwater model of a site can be used to estimate *a priori* the response of a natural flow system to the installation and operation of a GCW. As an interpretive tool, a well-constrained and calibrated groundwater model can be used to evaluate the performance of an existing system.

For the purposes of this protocol, and for the evaluation of GCW designs, it is recommended that a three-dimensional groundwater flow model be used to simulate the flow system of a site being evaluated for a GCW installation. Using a three-dimensional modeling approach, the area of influence of a GCW well may be examined in the upgradient and downgradient directions relative to the GCW as well as in the cross-gradient or lateral direction. Vertical flow velocities and travel times will also be of significance in the design phase of a GCW system.

Many existing groundwater flow modeling codes are currently available on the market. Table 12 contains a list of some of the three-dimensional (3-D) models commonly used for aquifer simulations. The intention of this protocol is not to endorse a specific code, but to suggest a non-proprietary code (that may also be provided privately) that will serve as an example of the type of modeling code that should be used.

**Table 12. Selected Readily Available Three-Dimensional Groundwater Flow, Particle Tracking, and Contaminant Transport Codes.**

| Model | Description | Authors and/or Vendors |
|---|---|---|
| | | |
| **Groundwater Flow** | | |
| MODFLOW U.S. Geological Survey | 3-D finite-difference groundwater flow model; steady-state or transient. | McDonald and Harbaugh (1988) (public domain & multivendor availability) |
| **Particle Tracking** | | |
| MODPATH | Particle tracking program for computing 3-D path lines using steady-state output from MODFLOW. | Pollock (1989) (public domain & multivendor availability) |

| | | |
|---|---|---|
| PATH3D | Particle tracking program for calculating groundwater paths and travel times in steady-state or transient, 2- or 3-D flow fields. | Zheng (1989)<br>S.S. Papadopulos & Associate |
| **Contaminant Transport** | | |
| MT3D | 3-D solute transport model based on the method of characteristics (MOC) and modified method of characteristics (MMOC) technique. Can be used in conjunction with any block-centered finite-difference flow model such as MODFLOW | Zheng (1990)<br>S.S. Papadopulos & Associates |
| **Coupled Flow & Transport** | | |
| BIOF&T 3D | 3-D finite-element model for simulation of flow and transport in the saturated and unsaturated zones. Incorporates aerobic and anaerobic biodegradation kinetics. | Draper Aden Environmental Modeling, Inc. |
| FEMWATER | 3-D finite-element model for simulating density-dependent flow and transport. Incorporates previously developed 3D-FEMWATER (Yeh, 1987), a 3-D finite-element groundwater flow model for saturated and unsaturated media. | Lin *et al.* (1996)<br>U.S. Department of Defense Groundwater Modeling System (GMS)<br>Yeh (1987) |
| 3DFEMFAT | 3-D finite-element model of flow and transport through saturated-unsaturated media. | Scientific Software Group |
| HST3D | 3-D finite-difference model for simulating flow and associated heat and solute transport. | Kipp (1987)<br>(multivendor availability) |
| Waterloo Transport Code | 3-D finite-element groundwater flow and mass transport model | Waterloo Hydrogeologic Inc. |
| **Multiphase Transport** | | |
| STOMP - Subsurface Transport Over Multiple Phases | 3-D finite-difference unsaturated, saturated flow and multiple-phase transport model | White and Oostrom (1996) |
| **Modeling Environments** | | |
| Groundwater Modeling System (GMS) | Modeling environment incorporating MODFLOW, MODPATH, MT3D, and FEMWATER. | U.S. Department of Defense |
| ModelCad[386] | Pre- and post-processing environment for MODFLOW, | Geraghty & Miller, Inc. |

| | | |
|---|---|---|
| | MODPATH, MT3D, and MOC. | |
| Groundwater Vistas | Modeling environment incorporating MODFLOW and MODPATH. | Waterloo Hydrogeologic Inc. |
| Visual MODFLOW | Pre- and post-processing environment for MODFLOW, MT3D, PATH3D, MODPATH, etc. | Environmental Simulations Inc. |

Perhaps the most widely used and accepted groundwater modeling code is that of the U.S. Geological Survey (USGS), a modular, three-dimensional, finite-difference, groundwater flow model, commonly referred to as MODFLOW (McDonald and Harbaugh, 1988). MODFLOW simulates two-dimensional and quasi- or fully-three-dimensional, transient groundwater flow in anisotropic, heterogeneous, layered aquifer systems. MODFLOW calculates piezometric head distributions, flow rates, and water balances and includes modules for flow toward wells, through riverbeds, and into drains. Other modules handle evapotranspiration and recharge. Various textual and graphical pre- and post-processors are available on the market. Additional simulation modules are available through the authors and by third parties.

The results from MODFLOW can be used in particle tracking codes, such as MODPATH (Pollock, 1989) and PATH3D (Zheng, 1989), to calculate groundwater paths and travel times. MODPATH is a post-processing package to compute three-dimensional groundwater path lines based on the output from steady-state simulations obtained with the MODFLOW modeling code. MODPATH uses a semi-analytical, particle-tracking scheme, based on the assumption that each directional velocity component varies linearly within a grid cell in its own coordinate direction. PATH3D is a general particle-tracking program for calculating groundwater paths and travel times in transient three-dimensional flow fields. The program includes two major segments: a velocity interpolator, which converts hydraulic heads as generated by MODFLOW into a velocity field; and a fourth-order Runge-Kutta numerical solver with automatic time-step size adjustment for tracking the movement of fluid particles (U.S. EPA, 1993).

In addition to these codes, many groundwater flow modeling and contaminant transport codes are available. A comprehensive description of non-proprietary and proprietary flow and transport modeling codes can be found in the U.S. Environmental Protection Agency document entitled *Compilation of Ground-Water Models* (U.S. EPA, 1993). Depending on the project needs, the designer of a GCW system may want to apply a contaminant transport code that can utilize the calculated hydraulic head distribution and flow field from the flow modeling effort. If flow and transport in the vadose zone is of concern, use of a coupled or uncoupled, unsaturated/saturated flow and transport model should be considered.

Model construction consists primarily of converting the conceptual model into the input files for the numerical model. The hydrostratigraphic units defined in the conceptual model can be used to define the physical framework or grid mesh of the numerical model. In both finite-difference (such as MODFLOW) and finite-element models, a model grid is constructed to discretize the lateral and vertical space that the model is to represent. Stratigraphic units are represented in the model by layers that are defined by an array of grid cells. Each grid cell is defined by hydraulic parameters (hydraulic conductivity, storativity, cell thickness (cell top and bottom), etc. that control the flow of water through the cells.

Boundaries are simulated in the model by specifying boundary conditions that define the head or flux of water that occurs at the model grid boundaries or edges. Boundary conditions describe the interaction between the system being modeled and its surroundings. Boundary conditions are used to include the effects of the hydrogeologic system outside the area being modeled, while at the same time allowing the isolation of the desired model domain from the larger hydrogeologic system. Three types of boundary conditions generally are utilized to describe groundwater flow: specified-head (Dirichlet), specified-flux (Neumann), and head-dependent flux (Cauchy) (Anderson and Woessner, 1992). Internal boundaries or hydrologic stresses such as wells, rivers, drains, and recharge may also be simulated using these conditions.

**4.2.2 Pilot-Scale Testing.** It may be necessary to conduct a pilot test to verify the output from the hydraulic modeling exercise described above and to finalize the design of the GCW system. Pilot testing may not be required for situations where only one or two shallow GCWs are to be installed. For GCW systems that include two deeper wells or more than two shallow wells, pilot testing is recommended. The objective of the pilot tests is to eliminate uncertainties and avoid costly redesign.

For most engineering applications, pilot tests are often conducted at a reduced or pilot-scale. Tests are conducted to collect scale-up data for designing full-scale systems. Similar to designing GCW systems, pilot testing is conducted to collect data to validate and calibrate the model used in the preliminary design. The design of the full-scale GCW system is finalized using the calibrated model.

GCW pilot testing differs from most other engineering pilot testing in that a full-scale GCW is used, not a scaled down version. The test is termed "pilot" relative to the total number of GCWs in the final system design. Full-size GCWs must be used because the hydraulics of the well, the aquifer response, and the interactions of both, are dependent on the specifics of the well design (i.e., screen length, placement across stratigraphic layers, pumping rates, etc.). Pilot testing GCW systems includes installation of one or two GCWs and testing the aquifer response to pumping these wells.

A recent development for pilot testing GCWs is the dipole test. Dipole aquifer testing is a method that uses recirculation to obtain aquifer properties, (Kabala, 1993). The results give estimates on the vertical and horizontal hydraulic conductivities, sustainable pumping rates, infiltration rates, and zone of influence. Although this test is relatively new and its application is not well documented, it provides useful information for the design and operation of the GCW system.

Other aquifer testing methods that can be used for pilot-scale testing include constant rate discharge tests and/or slug testing. These tests utilize characterization wells, not a GCW. The characterization wells must be completed in both the anticipated extraction and injection zones, and each of these zones in both wells should then be tested independently using the other zones as observation points. The additional observation points are important because some of the more accurate test analyses require at least one observation point.

**4.2.3 Full-Scale Design.**

**4.2.3.1 Model Calibration and Execution.** Calibration of a groundwater flow model refers to the demonstration that the model is capable of producing field-measured heads and flows that are used as the calibration values or targets. Calibration is accomplished by finding a set of hydraulic parameters, boundary conditions, and stresses that when used in the model produce simulated heads and fluxes that match field-measured values within an acceptable range of error (Anderson and Woessner, 1992). Model calibration can be evaluated through the statistical comparison of field-measured and simulated conditions.

Often, model calibration is difficult because values for aquifer parameters and hydrologic stresses are typically known in relatively few locations, and their estimates are influenced by uncertainty. The uncertainty in a calibrated model and its input parameters can be evaluated through the performance of a sensitivity analysis in which the aquifer parameters, stresses, and boundary conditions are varied within an established range. The impact of these changes on the model output (or hydraulic heads) provides a measure of the uncertainty associated with the model parameters, stresses, and boundary conditions used in the model. It is important to calibrate with values that are consistent with the field-measured heads and hydraulic parameters to ensure a reasonable representation of the natural system. Calibration techniques and the uncertainty involved in model calibration are described in detail in Anderson and Woessner (1992).

After a model has been calibrated to observed conditions, it can be used for interpretive or predictive simulations. In a predictive simulation the parameters determined during calibration are used to predict the response of the flow system to future events, such as the operation of a GCW system. The predictive requirements of the model will determine the need for either a steady-state simulation or a transient simulation that would enable changing conditions and stresses over time. Model output (i.e., hydraulic heads) can be interpreted through the use of a contouring package, and can also be applied to particle tracking simulations to calculate groundwater pathways and travel times.

**4.2.3.2 Well Placement.** Completion of quality site characterization and preliminary design work enables goal-specific and precise GCW placement. After thoroughly characterizing plume dimensions, site hydrogeology, and contamination during the site characterization phase, the preliminary design work provides data about the size of each well's circulation cell, its expected treatment effectiveness, and expected well interactions. For sites that require source treatment and/or plume reduction, it is important that a sufficient number of wells be included in the system design to cover the zone of contamination. Circles delineating each well's circulation cell drawn directly over a plume on a topographic map illustrate this coverage. Well interaction data gathered during preliminary design work should help optimize well spacing. If the objective of the GCW system is to intercept a migrating plume, a sufficient number of wells will be needed to effectively capture and treat the entire width, and potentially the length of the plume as it passes through the GCW system. A thorough evaluation of any proposed well placement must be performed with the numerical flow model constructed during the preliminary design phase.

**4.2.3.3 System Component Selection.** It is the responsibility of the contracted GCW vendor to correctly select the appropriate components for the GCW system that will be implemented at any site. Selection and sizing of system components is GCW configuration specific. The components must be sized properly to operate the system according to the design specifications obtained from the modeling exercises described above.

Care must be taken to select well materials that resist physical and chemical deterioration. This is particularly true with the installation of a GCW because many groundwater contaminants accelerate degradation of well components. Well screens and casings must remain undamaged to accomplish the following:

- successfully restrict the passage of formation material into the well
- be structurally rigid enough to prevent collapse throughout the well's lifetime
- prevent clogging of the screen caused by corrosion and/or polymeric swelling.

Well screens and casings are available in a variety of materials, many of which are susceptible to specific groundwater contaminants and conditions. For instance, steel decays in corrosive environments; polyethylene is susceptible to aromatic and halogenated hydrocarbons; polyvinyl chloride (PVC) deteriorates in the presence of ketones, esters, and aromatic hydrocarbons; and polypropylene is subject to attack by aliphatic and aromatic hydrocarbons, and oxidizing acids (U.S. EPA, 1986b). Information on specific materials can be found in chemical resistance or compatibility tables. Vendors frequently make these tables available to prospective consumers by putting them in appendices of catalogues.

Composite wells contain combinations of materials that require special consideration when they consist of dissimilar metals. Dissimilar metals in direct contact with the soil are subject to accelerated corrosion due to a difference in potential between the metals (U.S. EPA, 1986b). Using a dielectric coupling to connect dissimilar sections will electrically isolate the metals and prevent this type of deterioration.

**4.2.3.4 Monitoring Network Design.** The three objectives of a GCW monitoring system are:

- monitor contaminant concentrations
  - across the well to assess in-well treatment effectiveness
  - in the circulation cell to evaluate in situ treatment effectiveness
  - outside the plume to monitor plume migration (point-of-compliance)
  - in the vadose zone to track fate and transport of vapors
- monitor hydraulic head
  - at the influent and effluent regions of the GCW to monitor head developed by the well
  - in the aquifer to monitor groundwater circulation
- monitor biodegradation in the aquifer and vadose zone

A number of monitoring devices can be used to meet the above objectives. Proper selection and placement of monitoring system components is important for evaluating GCW performance and is dependent on site- and application-specific requirements.

The installation of the following items is suggested to provide adequate data for evaluating the performance of a GCW system:

- sample access at influent and effluent of GCW
- in-well flow rate measurement device
- discrete-depth monitoring wells for gradient measurements (at least 3)
- discrete-depth groundwater monitoring points or wells for sampling

- point-of-compliance well downgradient from plume
- off-gas sampling access (if applicable)
- off-gas flow meter (if applicable)
- discrete-depth soil-gas monitoring points (if applicable)
- in situ flow meters (suggested if cost permits).

The above list should be considered minimum requirements, and if process-specific instrumentation is appropriate, it should also be installed.

A monitoring network that allows for groundwater monitoring within the zone of influence must be included in the design of a GCW system. The monitoring network should include nested monitoring wells that are designed with short (1 foot length) screened sections to allow collection of discrete samples. Piezometers can be installed to measure water levels around the treatment zone. Other in situ sensors are available that can measure in situ temperature or pressure and could be incorporated into the design of the GCW monitoring system. Use of specialized equipment such as down-well flow meters may require specific sized wells or other arrangements. Water sampling and level monitoring capabilities must be included in the GCW design to allow for sampling and monitoring immediately adjacent to the well.

GCW configurations that either incorporate SVE or vadose zone off-gas treatment should include a monitoring network for soil gas in the vadose zone. The network can consist of soil-gas probes from which whole soil-gas samples can be collected for analysis. Optional capabilities for monitoring temperature, pressure, and oxygen concentrations may be desired and can be included in the design. Thermocouples have been used successfully to monitor temperatures and the data can be collected using an automatic data logger or recorded using a Fluke™ meter. Typically type J or Type K thermocouples are used with the selection depending on the temperature range. System pressures can be monitored using pressure transducers that have a sensitivity of 0.01 inches of water. The data from these sensors also can be recorded using automatic data loggers or hand-held meters. In situ oxygen sensors are available and can be included in the design. These sensors are used to monitor oxygen concentrations in the vadose zone on a semi-continuous basis. The data from these sensors can be recorded with a data logger and downloaded onto a portable computer. The oxygen data can be used as an indicator of biological activity.

There are several considerations for placement of monitoring equipment. Monitoring system components should be placed both inside and outside the expected treatment zone. For plume interception, the most important placement is downgradient of the treatment system. This placement acts as the "point-of-compliance" location that monitors the contaminant concentration in the groundwater after it leaves the treatment zone. It will also be important to monitor in the upgradient direction. This will provide a baseline for the monitoring in order to remove the regional or natural effects on the hydrogeologic system, such as seasonal water-table fluctuations. Plume concentrations can also fluctuate because of an uneven distribution of contamination in the plume, and the concentrations entering the treatment zone will therefore need to be known. Additional positions can be placed cross-gradient, and within the treatment zone between the up- and downgradient positions. This should provide coverage for contaminant trends and pressure gradients around the treatment zone. At each well location around the treatment system, monitoring points should be positioned vertically in at least the extraction zone, and near the top of the water table. If there is a large separation of the extraction and injection screens, additional monitoring positions vertically may be advantageous.

## 5.0 GCW Installation Procedures

The effectiveness of GCWs, like other in situ remediation systems, is directly dependent on the quality of the installation procedures followed. System installation is planned and supervised by vendor personnel and documented with field logs, lithologic logs, well construction diagrams, and as-built diagrams for any treatment units. Meticulous documentation of well design and construction will provide an excellent resource in the event a problem is encountered. Problems that can be avoided by ensuring correct installation include, but are not limited to (1) leaking bentonite and/or grout seals, (2) plugged well screens, (3) leaking in-well packers, and (4) smeared borehole walls (reduced local conductivity). The selection of an appropriate drilling method and materials followed by careful well development and shakedown will help maximize GCW treatment effectiveness and prevent costly, time-consuming problems. The installation process must be supervised by experienced field engineers to ensure the GCW system components are fully functional, and that the installation component depths and observations are fully documented for later review.

### 5.1 Drilling and Well Completion

Boreholes for GCW placement should be drilled and completed according to vendor specifications to ensure adequate depth and diameter. The method of drilling will depend on the soil and formation characteristics, depth of installation, and vendor requirements. Some common drilling techniques used to install environmental remediation system wells include mud rotary, hollow-stem auger, and drive casing.

It is critical that the well be completed correctly to ensure proper GCW operation. Completion entails the placement of packing and sealant materials in the annulus of the borehole and concrete at the surface for structural support. The GCW vendor must provide qualified field personnel capable or properly installing the system. A detailed well construction diagram illustrating the depth and dimensions of the well, as well as the depths of all screen and sealant material added to the well annulus, must be completed during well installation. Careful and detailed field notes can be of immeasurable importance in later phases of the project.

**5.1.1 Drilling Methods.** Boreholes for GCW placement should be drilled according to vendor specifications to ensure adequate depth and diameter. The method of drilling will depend on the site-specific geological conditions, depth of installation, and vendor requirements. Some common drilling techniques used to install wells are outlined in the following sections.

**Hollow-Stem Continuous-Flight Auger.** Generally considered mobile, fast and relatively inexpensive, the hollow-stem continuous-flight auger is the most commonly employed drilling method in unconsolidated materials (U.S. EPA, 1986b). This method uses no drilling fluids, thus reducing the likelihood of decreased local hydraulic conductivity around the borehole. This method is not recommended for well depths exceeding 150 feet (U.S. EPA, 1986b).

**Solid-Stem Continuous-Flight Auger.** Unlike the hollow-stem auger, the solid-stem auger must be withdrawn from the borehole to insert well screens and casings. This restricts the usage of this technique to consolidated sediments or stable, fine-grained unconsolidated materials that are less prone to caving. In addition, this method cannot be used effectively in the saturated zone, thereby making it not feasible for the installation of a GCW.

**Drive Casing.** Used predominately in soft unconsolidated formations, drive casing drilling is simply the mechanical hammering of hardened metal casing into the formation. Materials inside the driven casing are removed. Because the borehole advances with the casing, no potential for caving exists even in the unstable of formations; however, heaving sands can present difficulties with the clean-out of the driven casing. When site conditions are favorable this is an effective drilling method for advancing large diameter boreholes.

**Rotary Drilling.** Rotary drilling bores a shaft with a rotating bit while the cuttings are continuously removed by circulation of air, water, or mud. The selection of the drilling fluid hinges on site-specific conditions. While air is well suited for use in hard-rock formations, it requires special consideration when volatile soil and groundwater contaminants are present to prevent exposing onsite workers. Using water may reduce the risk to onsite workers, but it hinders the recognition of water-bearing zones and can destabilize borehole walls leading to collapse. In contrast, muds composed of bentonite, barium sulfate, organic polymers, or polyacrylamides help solidify borehole walls and prevent caving. Unfortunately this solidification contributes to reduction of local hydraulic conductivity that must be restored by proper well development. In addition to reducing hydraulic conductivity, organic muds tend to stimulate microbial growth. This growth may either aid or hinder remediation efforts depending on the type of system installed and the nature of the contaminants.

**Cable-Tool Drilling.** Cable-tool drilling advances a borehole by repetitively lifting and dropping a heavy string of drilling tools (Johnson Division, 1975). This action, combined with the addition of water to the borehole, serves to crush the formation into a slurry so it can be pumped to the surface. Although comparatively slow, this method does offer some advantages in relatively shallow formations. For instance, it allows for the collection of excellent formation samples and detection of even relatively fine-grained permeable zones (U.S. EPA, 1986b). Use of this method has declined in recent years, making it difficult to find drillers that use it in some regions of the country.

**5.1.2 Packing and Sealant.** Once the desired well depth has been attained, and well screens and casings have been inserted downhole, fill materials must be added to the annular space between the borehole wall and well screens and casings. These materials may be poured down the borehole or, if a hollow-stem auger has been used, the materials are added between the casing and the inside of the auger stem as the auger is withdrawn. This process is sometimes referred to as well completion.

All screened intervals of the well must be surrounded by a chemically inert well-rounded material, such as quartz sand or gravel, to promote fluid flow to or from the well. This packing material is known collectively as the filter pack. Fabric filters are not recommended for use as packing material (U.S. EPA, 1986b).

The annulus between the screened sections must be packed with a sealant material, such as bentonite or grout, to avoid short-circuiting between the two screen sections. The sealant should have a permeability at least one or two orders of magnitude less than the surrounding formation (U.S. EPA, 1986b) and should prevent the formation of preferential paths to the surface or to another screened interval. Typically, a concrete well apron seals the uppermost section of the well and protects it from heaving above the frost line. Poor seals will allow fluid communication within the well borehole itself, greatly reducing the effect of the well on the surrounding soils. All materials should be added according to vendor specifications.

**5.2 Well Development**
Well developing is the process of removing fine-grained particles surrounding screened intervals by aggressively pumping the well with flow reversals and/or surges. For most GCW applications, it is extremely important that the wells be developed prior to long-term operation to prevent fines from being pumped into the well and plugging up the system. This is especially true for air lift systems that create a surging action during groundwater pumping. Flow reversals and surges induced by pumps, bailers, or surge blocks flush particles smaller than the well screen openings from the formation. Development must be more complete than is typical for monitoring wells; i.e., over pump until turbidity is low and the groundwater parameters are stable.

**5.3 Monitoring System Installation**
Monitoring devices must be installed so that high quality data can be collected over the life of the project. Monitoring system components are installed at depths that will provide sampling opportunities in 3 dimensions, so the circulation cell can be characterized. In addition to the drilling methods above, push techniques can be used to install monitoring equipment. Push installations do not require drilling and do not produce any cutting that may require treatment or disposal; however, pushing results in compaction of the adjacent soil, which could interfere with measurements. Either a hydraulic or pneumatic push technique can be used to install monitoring equipment. Pneumatic techniques utilize a hammering action, so equipment sensitive to shock or vibration should be installed with the hydraulic push technique.

In situ oxygen sensors (for vadose zone monitoring only), flow meters, and thermocouples should be installed according to manufacturer's specifications. Typically, accurate monitoring with these units requires surrounding material to emulate local sediments. Some sensor types can be installed in the same borehole as monitoring points to save costs. Some more specialized devices require separate installation so that there is no interference from other sensing devices. Typically, hydraulic push drilling methods are inappropriate for installing sensitive electrical or electrochemical sensors.

**5.4 System Shakedown**
After the installation of all GCW system components is completed, the system should be operated vigorously to challenge system components and reveal weaknesses or failures. This shakedown procedure should last long enough to ensure that the system integrity will allow long-term operation with minimal downtime. During shakedown, pumps, blowers, and/or compressors should be run at maximum required pressures and flow rates. System components and machinery should be carefully inspected for leaks; loose fittings; unusual vibration, noise, or movement; excessive temperatures; and other failures. All monitoring system components should be checked to make sure that they operate properly. Any defects should be fixed and their repair should be recorded. Beyond system repair, the cause of any failure should be investigated to ensure that proper preventive measures have been performed to avert future failures. Once the shakedown procedure is completed, the system is readied for long-term operation.

## 6.0 GCW Performance Monitoring Requirements

Monitoring is required to optimize the operation of the equipment and to verify GCW system performance. In well, aquifer, and vadose zone monitoring must be conducted to collect the data required to determine any needed system adjustments, and to assess the effectiveness of the GCW for removing the desired contaminants. The information required to evaluate and optimize a GCW system includes the following:

- the amount of contaminant mass removed or destroyed
- the zone of influence around the well
- the amount of groundwater circulated
- the potential secondary effects of operating the wells.

Performance monitoring is the periodic collection of data that provides information necessary to evaluate the status of the operating GCW system and the surrounding aquifer. Monitoring is essential to optimize GCW system performance, detect impending failures and unwanted effects, and to demonstrate removal of contaminants from the site. Without monitoring, a GCW system could continually function significantly below its potential removal and economic efficiencies, wasting both time and money. While there are significant costs associated with performance monitoring, those costs are easily recovered by optimizing energy and removal efficiencies, and reliably documenting cumulative contaminant mass removal.

This section describes methods and technologies used to monitor the performance of GCW systems. Monitoring not only optimizes GCW operation, but also verifies system contaminant removal performance. Monitoring will be used to determine the following:

- general status of the system processes
- amount of contaminant mass removed or destroyed in the GCW
- reductions in aquifer contaminant concentrations
- zone of influence around the well
- amount of groundwater circulated
- potential secondary effects of operating the wells.

The specific parameters measured, however, will be determined in part by the treatment process (chemical, physical, biological) of the selected GCW design. For example, if a bioprocess is being considered, respiration gases must be monitored to provide evidence for treatment process effectiveness. Conversely, monitoring respiration gases in a GCW system using air-stripping and an aboveground carbon adsorption unit to treat the off-gas would not prove useful.

### 6.1 In-Well Monitoring Requirements

The monitoring of processes that occur within the GCW is important for determining the system's effectiveness, adjusting operating parameters, and predicting the size of the circulation cell and required treatment time. In-well monitoring parameters include water flow rates (groundwater pumping rates), hydraulic head measurements, vapor flow rates, contaminant concentrations in the influent and effluent waters, and biological parameters. These measurements are required for mass removal calculations and can also be compared to the predicted values generated during the modeling and design phases of the project as a system performance check.

For any GCW configuration, the in-well flow rates must be known to perform mass removal calculations and are important for process and operational performance monitoring. The groundwater-pumping rate refers to the volume of water passing through the well in a unit of time. Hydraulic heads are measured at the well to determine the gradient imposed on the aquifer by the GCW. The vapor flow rate is important for systems that utilize air lift, air stripping, and/or SVE. The flow rates are needed to perform mass balances on contaminants.

The performance of the in-well treatment process is monitored to evaluate the system's performance, and predict the time that will be required to achieve the remediation goals. Treatment efficiency is determined by comparison of influent and effluent contaminant concentrations.

It is important to monitor parameters that affect biological activity in systems that incorporate in-well biological treatment or that are designed to deliver compounds to support in situ biodegradation.

**6.2 Aquifer Monitoring Requirements**
Aquifer monitoring is conducted to evaluate groundwater circulation imposed by the GCW, to assess the mass of contaminant removed, and to evaluate biological processes occurring in the aquifer. Hydraulic head measurements at locations around the GCW are used to evaluate the movement of groundwater. Analyzing groundwater samples provides data on contaminant removal, migration, and/or biological processes.

**6.2.1 Groundwater Sampling and Analysis.** Groundwater samples can be collected from monitoring wells, remediation/recovery wells, or production wells. Monitoring wells, which are specifically designed and controlled for purposes of sampling, yield the most reliable and representative groundwater samples. The groundwater analyses are necessary to define or estimate the nature and extent of groundwater contamination. Determination of the plume configuration will help to optimize recovery well placement.

The groundwater wells, depending on the screened interval, can be used to determine whether more than one phase of fluid is present in the aquifer. Light non-aqueous phase liquids float on the water table; DNAPLs sink to the bottom of the aquifer but can be trapped in pockets throughout the saturated zone. Some LNAPLs and DNAPLs may contain emulsifiers, may emulsify naturally in the groundwater, or may be emulsified by other contaminants in the groundwater. In these instances, the contact between the phases may be indistinct and recovery or observation of the free phase may be affected.

Groundwater samples are collected from all available wells on and around the site. New well installations must be developed prior to sampling by pumping out several well volumes of water to ensure that the collected water is representative of the aquifer in the region of the well. The number and frequency of groundwater sampling events for characterization purposes should be in inverse proportion to the quality and quantity of groundwater data collected at the site and discovered in the background investigation. If little is known about the concentration and location of the contaminated groundwater, further sampling will be required to characterize the contaminant plume. The money spent characterizing a site is easily recovered over the life of a project, within reason.

Collected samples are analyzed for the contaminant of interest, as well as any other water parameter that may affect either the current contaminant concentration or the effectiveness of the GCW, once operations commence. For example, if a degradation product of a primary contaminant has been determined to inhibit or compete with the degradation of the parent compound, then the daughter species must be characterized initially and monitored (along with the primary contaminant) throughout the project.

**6.2.2 Water Level/Pressure Measurements.** The water level in the GCW, in piezometers placed adjacent to the influent and effluent screen sections of the GCW system, and in monitoring wells placed around the zone of influence should be monitored to evaluate the ROI of the GCW. Although the pumping drawdown is offset by the recirculation of water from the well, water level and pressure head differences are observable and are useful for monitoring the performance of pumping in the well and the ROI in the formation.

A variety of tools are available to measure water levels in GCWs and monitoring wells. These include wetted tape, electric tape, electronic pressure transducers, sounding (acoustical) devices, oil/water interface probes, and airline pressure apparatus. A mechanical water level chart recorder can be installed to measure relative changes in the water level for a predetermined time interval. This can prove useful at system startup during the dynamic phase of operation. Pressure transducers can be installed in a network of monitoring wells to provide continuous measuring of water levels. The output from the transducers can be recorded and transmitted or saved by a data-logger, then downloaded to an electronic file in the field.

Pressure transducers can be difficult to set in the pumping zone, as it is necessary to feed the transducer lines through or along side the treatment unit inside the circulating well. Problems can arise when trying to feed these lines around packers or other in-well components. Turbulence in airlift GCW systems could interfere with pressure transducer readings. Incorporating a stilling tube into the well design could overcome this problem.

**6.2.3 Radius of Influence Measurements.** The aquifer area that can be effectively remediated by one GCW unit can be defined by the distance to which the required level of treatment can be imparted within a feasible project life. This distance is commonly referred to as the ROI or sometimes the effective ROI. Another term commonly used to describe the outer distance of GCW effectiveness is the stagnation point. ROI will be the term used in this document for the purpose of discussion.

There is some challenge in defining the ROI. For example, with GCW configurations that incorporate in-well treatment, the contaminant must be transported to the well as the groundwater is drawn to the well. The process relies on desorption and "flushing" of contaminant as water circulates in the aquifer. It may take many cycles for the flushing mechanism to reduce the contaminant concentrations to targeted levels. The ROI for this GCW configuration is the distance at which the number of groundwater cycles is sufficient to achieve the remediation goals within the desired project duration.

When GCW systems are used for plume interception, the ROI is defined as the distance from the well at which the plume is effectively "captured" into the circulation cell. Determining the ROI requires the application of a groundwater model. Using the design pumping rate, hydraulic gradients and groundwater velocities are calculated to determine how the background groundwater flow will affect the dimensions of the GCW circulation. Knowing this achievable

ROI is extremely important for proper placement of the wells to ensure that the plume does not "pass by" the treatment system.

For GCW configurations that utilize the aquifer as an in situ bioreactor, the ROI is defined as the distance to which nutrients and/or electron donors or acceptors can be delivered. The ROI depends on the rate of delivery, the rate of sorption and desorption, the rate of decomposition, and the rate of microbial and/or abiotic utilization. Typically, the distance from the well that nutrients or electron donors can be delivered is small because of the high demand just outside the well. As the contamination near the well is degraded, the demand will decrease and distance of delivery will increase.

In summary, the definition of ROI with regard to GCW technology, as well as appropriate monitoring methods, are dependent on the system configuration, treatment technologies, and system objectives. A site-specific operational definition of ROI is suggested.

The following sections will present various GCW ROI monitoring techniques that can be applied to appropriate GCW configurations. The specific method used to evaluate the in situ performance of a GCW system must be matched to the mass transfer and removal technologies employed by that system.

**6.2.3.1 Dye/Tracer Tests.** Groundwater circulating well systems and sites with minimal aquifer monitoring installations may not produce adequate data to evaluate the GCW ROI or circulation cell. Systems with low pumping rates may induce only a very shallow head gradient around the GCW, making it difficult to determine the circulation radius. Head and water level measurements taken in monitoring wells may be insufficient to characterize the treatment area. Therefore, tracer or dye testing is required to determine the actual ROI. These tests also can be used to compare and confirm head measurements from sites with plenty of monitoring installations.

Tracer and dye tests are performed to determine groundwater flow paths and velocities in the aquifer surrounding a GCW. Two distinct tracer test approaches include the divergent test and the convergent test. The divergent test is conducted by injecting a tracer compound or dye into the GCW and then periodically collecting groundwater samples from monitoring wells or points at different depths and distances from the GCW. The data are used to determine the travel time of the tracer from the GCW to the monitoring locations.

With the convergent approach, a dye or tracer is injected into the groundwater via monitoring wells or injection points at some distance from the GCW. The GCW influent is monitored for arrival of the tracer. It should be noted that regulations pertaining to the injection of materials into groundwater vary and should be investigated thoroughly to ensure compliance. The selection of the appropriate tracer and tracer testing methods is specific to the contaminant characteristics and GCW design, therefore, this selection should be performed on a site-specific basis.

Tracers are natural or introduced components to a groundwater system that can provide information on the movement of the groundwater. Tracers can be thermal (water temperature), or can consist of particles such as biological solids (yeast, bacteria, spores), ions, organic acids, dyes, and radioactive isotopes. The selection of a particular tracer is dependent on the data objectives and on the natural system into which the tracer will be introduced. The ideal tracer would accomplish the following:

- not interact with the aquifer matrix
- not be affected chemically by the groundwater
- be non-toxic
- move only at the rate of groundwater movement
- be easily detected
- have similar physical and/or chemical properties to the material being traced.

For instance, sulfur hexafluoride ($SF_6$) is used to mimic oxygen transport in aquifers because its diffusion coefficient is similar to the diffusion coefficient of oxygen.

Inorganic ionic compounds such as chloride, bromide, sulfate, and iodide; and organic compounds such as fluorobenzoate have been used extensively as groundwater tracers. The inorganic ionic compounds are easily detected in the field using ion-specific electrodes, or by measuring changes in the groundwater-specific conductance (electrical conductivity) as the tracer moves past a monitoring point. The disadvantages of ions include chemical dispersion, interference from background ions, and sorption by microorganisms or soil particles.

Various organic, fluorescent dyes are also widely used ground water tracers. Fluorescent dyes are relatively inexpensive to use and are easily detected. Natural groundwater factors such as suspended solids, hardness, salinity, pH, and temperature can interfere with the movement and detection of the dyes. Both ionic detectors and fluorimeters (for quantification of fluorescent dyes) can be installed in a well or can be operated at ground surface to analyze collected samples.

### 6.3 Vadose Zone Monitoring Requirements

Vadose zone monitoring is important for assessing fate and transport of vapors with GCWs that incorporate SVE or direct injection of their off-gas. Vadose zone monitoring consists of collecting soil-gas samples and analyzing for contaminant concentrations and/or respiratory gases. Pressure monitoring can be used to determine the ROI of the GCW in the vadose zone.

**6.3.1 Soil Sampling and Analysis.** Soil borings are located based on either the review of existing site data or the results of the soil-gas survey. Soil borings can serve two purposes: the collection of soil samples and the installation of circulating wells and monitoring points. Soil borings have the advantage of allowing a large number of soil samples to be collected from a single location and allowing for subsequent installation of the circulating wells and monitoring points in the borings. Disadvantages include the generation of soil cuttings and the fact that drilling may require subcontracting and a large amount of time. Alternative methods, such as a GeoProbe™ system or cone penetrometer, may be used for collection of soil samples and may be suitable for installing soil-gas monitoring points.

The hollow stem auger method is generally preferred for drilling in unconsolidated soils; however, a solid stem auger is acceptable in more cohesive soils. The final diameter of the borehole is dependent on the diameter selected for the circulating wells, but typically it is at least two times greater than the outside diameter of the circulating well.

All drilling and sample collection activities must be observed and recorded on a geologic boring log. Data to be recorded includes soil sample interval, sample recovery, visual presence or absence of contamination, soil description, and lithology. Soil samples must be labeled and properly stored immediately after collection.

It is preferable that all boreholes be completed as circulating wells or monitoring points. If this is not possible, boreholes must be abandoned according to applicable state or federal regulations. Typically, borehole abandonment is accomplished by backfilling with bentonite or grout.

The soil samples are analyzed for appropriate contaminants. Additionally, moisture content, particle size, and total Kjeldahl nitrogen are useful in characterizing the soil and biodegrading potential. A summary of some soil analyses that might be appropriate is provided in Table 13.

**Table 13 Analyses and Appropriate Methods for Site Characterization.**

| Analysis | Method |
|---|---|
| Aromatic hydrocarbons (BTEX) | Purge and trap GC method SW8020 |
| TPH | Modified GC method SW8015 |
| Moisture Content | ASTM D-2216 |
| Particle Size Analysis | ASTM D422 |
| Total Kjeldahl Nitrogen | EPA 351.4 |

**6.3.2 Soil Gas Sapling and Analysis.** Soil gas samples are collected from the monitoring probes in the vadose zone and analyzed for oxygen, carbon dioxide, and the contaminant of interest. The sampling method described *A Field Test for Bioventing* can be used to sample soil gas for GCW applications (Ong et al., 1994). Analysis for TPH can be accomplished in the field using hand held meters, more specific compound analysis may require laboratory analysis.

**6.3.3 Surface Emission Testing.** Surface emission tests are required with systems that discharge their off-gas to the vadose zone for treatment. The tests are conducted to quantify the mass flux of contaminant emissions from the ground surface to the atmosphere resulting from operation of a GCW system. The mass flux of contaminant vapors leaving the ground surface is quantified to accomplish the following:

- track the fate of vapor-phase contaminants
- calculate removal efficiency of the system
- comply with regulatory limits on vapors released to the atmosphere.

Several methods have been reported to be effective in quantifying surface emission mass fluxes (DuPont, 1987; Pollack and Gordon, 1993). Typically, these tests quantify the mass of contaminant being released from a defined area within a defined time period. Contaminant mass flux rates are described in terms of units of mass per unit area per unit time (mg contaminant/$m^2$-day). Samples are collected with the GCW system running and turned off. The increase in mass flux from background (system off) is the contribution from the operation of the GCW.

Performing surface emissions tests typically involves passing a pure air stream slowly over the ground surface under a box made of inert material. The air stream is collected at the downstream end of the box for analysis. Evacuated Summa™ canisters or sorbent materials may be used as storage devices until the air samples can be analyzed, usually by GC. The total mass of contaminant in the sample, is related to the time and ground surface area over which the sample is collected, to yield the mass flux rate of contaminants emitted to the atmosphere.

### 6.4 GCW Off-Gas Monitoring Requirements and Procedures

System off gas monitoring is required to ensure compliance with regulatory discharge permits and to perform mass balance calculations to monitor GCW performance. Monitoring for regulatory purposes involves measuring the off-gas flow rate and collecting gas samples from the discharge of the off-gas treatment system. The required sampling frequency is location dependent and must by agreed upon with the acting regulatory agency during the planning stage. The concentration of targeted contaminants is measured in the off-gas samples using appropriate and approved analytical methods. The results are used to calculate a mass discharge rate by multiplying the concentration by the flow rate. It is necessary to ensure that the mass discharge rate does not exceed the permitted discharge rate. If it does, it will be necessary to halt operations until the system is brought into compliance.

GCW performance monitoring requires sampling the off-gas from the well before the off-gas treatment system. Samples can be collected in Tedlar™ bags using the techniques described in Section 3.3.2, or in evacuated Summa Canisters. Samples are collected more frequently at startup, then less frequently as the system operates and approaches steady state conditions. At a minimum off-gas samples should be collected daily for the first two weeks, then twice per week for the next six weeks, then less frequently depending on the trends observed in the contaminant concentrations. The data are required to complete the mass balance around the GCW and to verify that the system is achieving the design stripping efficiency.

The data from the well off-gas samples also are used in conjunction with the off-gas data from the off-gas treatment system to monitor the effectiveness and efficiency of off-gas treatment. Concentrations in the well off-gas must be monitored to determine if the appropriate off-gas treatment technology is being used and/or when it should be changed out.

### 6.5 GCW Performance Modeling.

The data from the above monitoring is compared against the predictions of the system model. The model is validated based on the fit or agreement between the predicted system performance and the performance measured in the field. If necessary, the model is calibrated to more accurately predict future system performance.

## 7.0 GCW Performance Assessment

The performance of a GCW system involves assessing the contaminant removal within the well system, and the mass reduction in the aquifer and when appropriate the vadose zone.

### 7.1 In-Well Contaminant Mass Removal Rate

The effectiveness of the GCW is assessed based on the contaminant mass removal rate and efficiency. The rate of contaminant mass removal dictates the time required for achieving cleanup goals and indicates the cost-effectiveness of the system. Assessing the effectiveness of the GCW technology entails knowing the groundwater pumping rate and the influent, effluent and off-gas contaminant concentrations. A mass balance is constructed as follows:

$$\text{Conc}_{\text{inf}} \times \text{FR}_{\text{inf}} = \text{Conc}_{\text{eff}} \times \text{FR}_{\text{eff}} + \text{Conc}_{\text{off-gas}} \times \text{FR}_{\text{off-gas}} \quad (3)$$

where: $Conc_{inf}$ = contaminant concentration in the water entering the well
$Conc_{eff}$ = contaminant concentration in the water leaving the well
$FR_{inf}$ = flow rate of water entering the well
$FR_{eff}$ = flow rate of water leaving the well
$Conc_{off\text{-}gas}$ = concentration in the off-gas from the well
$FR_{off\text{-}gas}$ = off-gas flow rate

If proper sampling and analytical procedures were followed, the mass balance should close. If the mass balance is off by more than 15%, the data are suspect and accurately assessing well performance will not be possible. Often times, the problem with poor mass balance performance is associated with collecting non-representative samples of the well influent. Sampling at one point within the annulus of the well will not provide a representative sample. It is much better to sample from within the well after the water has had a chance to mix but before any in-well treatment occurs.

Provided that the results of the mass balance are acceptable and, the mass removal rate and the stripping efficiency can be calculated. The mass removal rate is the mass of contaminant that is removed from groundwater pumped through the well per unit of time. It is calculated as follows:

$$MRR = (Conc_{inf} - Conc_{eff}) \text{ x } PR \quad (4)$$

Where: MRR = mass removal rate
PR = pumping rate

In-well mass removal rates are useful for predicting the ROI (using modeling) and estimating the percentage of water circulated through the well from within that radius.

The treatment efficiency is a measure of the effectiveness of the well system for removing contaminant from the groundwater. It is calculated as follows:

$$\text{Efficiency} = \frac{[(Conc_{inf} \times FR_{inf})] - [(Conc_{eff} \times FR_{eff})]}{[(Conc_{inf} \times FR_{inf})]} \times 100 \quad (5)$$

With most GCW systems the influent flow rate equals the effluent flow rate so equation 5 reduces to:

$$\text{Efficiency} = \frac{Conc_{inf} - Conc_{eff}}{Conc_{inf}} \times 100 \quad (6)$$

While the stripping efficiency of GCW systems cannot equal the stripping efficiency of an above ground air stripping reactor, typical values of 75% to 90% are common. Stripping efficiencies much below 75% indicate that the well system has been poorly designed and/or operated.

Another operational parameter used to assess GCW performance is the air to water ratio. This is simply the volume of air injected to the volume of groundwater pumped. Typically, higher air to water ratios result in greater removal of contaminant. With GCW systems, the air to water ratio is limited due to the size of the air stripping reactor and the requirements for air lift pumping. Air to water ratios have been reported in the range from below 10 to over 200. The goal for optimizing GCW operation is to balance the air to water ration and the stripping efficiency and maintain a sufficient groundwater-pumping rate.

**7.2 In Situ Contaminant Concentration Reduction-**
The most critical measure of GCW operation is the reduction in mass of contaminant within the volume being treated. This includes both the vadose and saturated zones. The mass of contaminant removed from the vadose and saturated zones should include the contaminant removed within the GCW and any contaminant that was biodegraded or removed by other abiotic mechanisms.

**7.2.1 Vadose Zone Soils.** Often, GCW systems are designed to simultaneously remove contaminants from both the vadose and saturated zones. This can be accomplished by either incorporating a SVE component in the GCW design or by injecting the off-gas into the vadose zone to support bioventing and biodegradation. The following data are required to determine the mass of contaminant removed from the vadose zone.

- contaminant concentrations measured in soil samples collected before and following GCW operation.
- the ROI of the GCW in the vadose zone
- the thickness of the vadose zone
- the bulk density of the soil within the treatment volume

The data are used to determine the mass of contaminant in the vadose zone before and following GCW operation. The mass removed is simply the difference between the two resulting values.

Determining the mass of contaminant within a soil volume based on the measured concentrations can be tedious. The use of computer programs such as EarthVisions™ makes this task easier. This program generates a 3-D graphical representation of the contaminant distribution. The concentration ranges are broken down into shells. The program can calculate the volume of contaminated soil that falls within each shell. The mass of contaminant can be calculated by taking the average concentration of a shell, multiplying it by the volume of the shell, then multiplying the result by the bulk density of the soil.

**7.2.2 Saturated Zone Soils.** Mass reductions of contaminant in saturated zone soils are calculated in much the same way as the vadose zone soils. The trick is that the analyses require that the mass contributions from the interstitial water must be taken into account. The following data is required to determine the mass of contaminant removed by a GCW system.

- contaminant concentrations in saturated zone soil samples before and following GCW operation
- water content of the saturated zone soil samples
- contaminant concentrations in interstitial water

To do this, the water content of the sample must be determined. Then an aliquot of interstitial water is removed and analyzed for the contaminant(s) of interest. Finally, the soil including the interstitial water is analyzed. The actual soil concentration is calculated by determining the volume of water within in a unit volume of the soil, multiplying the result by the concentration, then subtracting the result from the measured soil concentration. Once the soil concentrations, the data are used to calculate mass removal using the procedure described for vadose zone soils. Because determining contaminant concentrations in saturated soils is time consuming an costly, performance of more commonly assessed based on reductions in the concentrations and mass of contaminant in groundwater.

**7.2.3 Groundwater.** The more common method for assessing the performance of an aquifer restoration technology is to monitor decreases in contaminant concentrations in groundwater. The following data is required to perform this analysis at sites where the groundwater velocity is very low.

- contaminant concentrations in groundwater within the circulation cell before and following GCW operation
- volume of the circulation cell
- porosity within the circulation cell

The contaminant mass is calculated using an approach that is similar to the approach for the vadose and saturated zone soils. First, the concentration data is entered into a computer program that generates a 3-D profile of the contaminant distribution. Nest, the volumes of concentration shells are calculated. The total mass of contaminant is calculated by taking the average concentration in each shell and multiplying it by the volume of that shell, then multiplying by the porosity and adding the masses for all of the shells. The mass removal is calculated as the total mass before GCW operation minus the mass remaining after GCW operation.

If a significant quantity of groundwater moves past the GCW system, it is more appropriate to calculate the mass removed based on the mass flux of contaminant that enters and leaves the circulation cell. The following data are needed to make this calculation.

- flux of water into and out of the circulation cell during GCW operation
- contaminant concentrations in water entering and leaving the circulation cell during GCW operation
- contaminant concentrations in groundwater within the circulation cell before and following GCW operation
- volume of the circulation cell
- porosity within the circulation cell

The mass removed from within the circulation cell can be determined as described above. The additional mass removed from the groundwater passing through the GCW system is calculated as the difference between the contaminant concentration in the water entering the circulation cell and the contaminant concentration leaving the circulation cell multiplied by the volume of water that passed through the circulation cell.

### 7.3 Dilution Effect in the Circulation Cell

Both in-well treatment efficiency and dilution effects should be considered when estimating the total effectiveness of the system on the surrounding plume. Because water circulating near the well is likely to be treated repeatedly, it will have a relatively low contaminant concentration. The contaminated groundwater entering the circulation zone mixes with this relatively clean water before it enters the GCW. As a result, the contaminant concentration of groundwater entering the GCW circulation zone will be lowered by dilution.

Comparing contaminant concentrations in samples collected from the GCW influent screen with samples taken at groundwater monitoring locations nearby provides information about dilution effects. For example, if the concentration in the regional plume outside the circulation cell is 100 mg/L, and the concentration at the influent screen is 1 mg/L, there is a 100-fold dilution of contaminant concentration in the circulation cell before the plume water reaches the GCW. This clearly assumes that no removal is being performed in the aquifer sediments themselves. The dilution factor is a function of the rate at which contaminated groundwater enters the circulation cell and the volume of water in the circulation cell. If the dilution factor can be quantified and one of these variables is known, the other variable can be determined. Information such as ambient groundwater velocity and concentrations from all groundwater-monitoring installations will be useful for estimating these variables.

The circulation cell around the GCW will contain groundwater that varies in contaminant concentration. Characterizing the concentration gradient between the cleaner water around the GCW and the unaffected contaminant plume may be problematic, thus preventing accurate estimation of the circulation cell volume and/or the ROI. Nevertheless, the dilution factor should be monitored and documented throughout the operational life of the GCW. Changes in the dilution factor may indicate any of the following:

- changes in circulation cell volume
- changes in ambient groundwater velocity
- changes in aquifer hydraulic conductivity
- establishment of removal processes in aquifer sediments.

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