3-2002

An Analysis of Horizontal Flow Treatment Well Applicability for the Treatment of Chlorinated Solvent Contaminated Groundwater at United States Forces Korea Installations

Michael R. Staples

AN ANALYSIS OF HORIZONTAL FLOW TREATMENT WELL APPLICABILITY FOR THE TREATMENT OF CHLORINATED SOLVENT CONTAMINATED GROUNDWATER AT UNITED STATES FORCES KOREA INSTALLATIONS

THESIS

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AFIT/GEE/ENV/02M-14

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THESIS

Presented to the Faculty

Department of Systems and Engineering Management

Graduate School of Engineering and Management

Air Force Institute of Technology

Air University

Air Education and Training Command

In Partial Fulfillment of the Requirements for the

Degree of Master of Science in Engineering and Environmental Management

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March 2002

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Acknowledgements

There are many individuals whom I'd like to recognize for helping me complete this thesis. First, I would like to thank my thesis advisor, Dr. Mark Goltz. His foresight, guidance, and patience made this thesis the product it is now. His insight and ability spared me a great deal of time and frustration. I would also like to thank Dr. Junqi Huang for the many hours he spent explaining MODFLOW and the Fortran code of the HFTW model.

The most important contributor to this these effort is my wife. I am extremely appreciative of all the support and encouragement she gave me. There were many hours that I was in the house but she was alone, her patience and understanding will be remembered by me always.

Michael Staples

Table of Contents

Page

Page

List of Figures

Figure Page

Page

Page

List of Tables

AFIT/GEE/ENV/02M-14

Abstract

Past research (Oshiba, 1997) has shown that there is a rising public concern with environmental issues in the Republic of Korea (ROK). As Korean government and public interest in the environment grow, there is likely to be increased pressure to remediate environmental contamination at United States Department of Defense (DoD) installations in Korea. Impacting DoD's ability to remediate contaminated sites overseas is the fact that limited environmental funds must compete with high priority mission requirements. Thus, particularly at overseas bases, there is an urgent need for inexpensive and effective groundwater remediation technologies.

This study focused on the containment of groundwater contaminated with chlorinated solvents in the fractured rock aquifers that are commonly encountered at DoD installations in the ROK. Horizontal Flow Treatment wells (HFTWs) were analyzed as a potentially cheaper, safer, and more effective technology for the containment of chlorinated solvent contaminated groundwater. Both hydrogeologic and design parameters were varied to determine their effects on the technology performance.

From this study, it was determined that an HFTW numerical model developed for porous media is appropriate for application in the fractured systems encountered in the ROK, and that HFTWs have the potential to be a cost effective alternative for contaminant management in fractured media when compared to conventional technologies. Model analysis indicated HFTWs might be appropriate for containing contaminant plumes in the ROK, though bypassing of system may be problematic.

AN ANALYSIS OF HORIZONTAL FLOW TREATMENT WELL APPLICABILITY FOR THE REMEDIATION OF CHLORINATED SOLVENT CONTAMINATED GROUNDWATER AT UNITED STATES FORCES KOREA INSTALLATIONS

1 Introduction

1.1 Motivation

Past research has shown that there is rising public concern with environmental issues in the Republic of Korea (ROK) and that as Korean government and public interest in the environment grow, there is likely to be increased pressure to remediate environmental contamination at United States Forces Korea (USFK) installations (Oshiba, 1997). A failure by the US to be viewed as a good steward of the environment by the Korean government and public could eventually threaten US access to the land, air, and sea resources in the ROK that are critical to the success of US defense policy. In fact, one point that has repeatedly been made by Korean groups protesting the US military in Korea has been the degradation of the environment that has resulted from the US presence (Kang, 2001; Lee, 2001a; b; Shin, 2001b; Son, 2001). While the current mutual defense treaty and Status of Forces Agreement (SOFA) between the US and ROK governments do not require remediation of environmental contamination at USFK installations, changes in US remediation policy were seen in recent negotiations, which amended the SOFA, in part to address environmental pollution around USFK bases (US Embassy Tokyo, 2000; DoD, 2001; Lea, 2001).

The newly agreed to SOFA required the US and ROK to sign a memorandum of special understanding to include cooperative measures for environmental protection (US

Embassy Tokyo, 2000; Lea, 2001). Under the new memorandum, the United States is required, among other things, to review and update Environmental Governing Standards biennially for the purpose of accommodating more protective rules and standards, consult with the ROK on any risks posed by environmental contamination on USFK facilities or in communities adjacent to such facilities, and promptly undertake actions to remedy contamination caused by United States Armed Forced in Korea that poses a known, imminent and substantial endangerment to human health (DoD, 2001).

The SOFA and memorandum of special understanding negotiations came after precedents were set at other US military installations overseas where the US funded remediation efforts or paid substantial penalties for returning contaminated installations to the host nations (Oshiba, 1997). The latest precedent came on 17 May 2001, when the House of Representatives passed an amendment to the Foreign Relations Authorization Act for Fiscal Years 2002 and 2003 that supports an assessment to examine environmental problems resulting from former US military facilities in the Philippines (Brooks, 2001; Gault, 2001). These precedents seem to indicate that it may just be a matter of time before the US DoD becomes responsible for remediation of environmental contamination at its ROK installations.

Outlining actions necessary for the turnover of US facilities to a foreign government, AFI 32-7006 states, "Depending on the terms of the governing international agreement, environmental cleanup costs for US caused environmental contamination may be included in the host nation's overall damage claim." (AF, 1994) The amended SOFA does not include the requirement for cleanup or compensation in situations other than those that pose an imminent threat to human health, but the memorandum includes provisions to change this. Such a change in remediation requirements could pose substantial budgetary impacts to USFK installations.

Past research has found evidence of chlorinated solvent spills penetrating aquifers and contaminating groundwater at USFK installations (Hartman, 1999). Contaminants dissolved in groundwater can be transported to both environmental and human receptors. Chlorinated solvents such as trichloroethylene (TCE) and tetrachloroethylene (PCE), and their degradation products, are characterized as possible carcinogens by the Environmental Protection Agency (EPA, 1995b). These solvents have maximum contaminant levels (MCLs) established by the EPA under the Safe Drinking Water Act. These MCLs specify the maximum concentration levels of the contaminants that are acceptable in drinking water. At Osan AB in the ROK, TCE has been detected in groundwater wells at concentrations of 83 μg/L, 17 times its MCL, and vinyl chloride at concentrations of 20 μg/L, ten times its MCL (Osan AB CE, 2001). In the event of contingency operations and a massive build-up of manpower at USFK bases, these wells may be utilized to provide drinking water to US troops. Additionally, groundwater could transport these contaminants off USFK installations, polluting civilian water sources. There is therefore a need to control the spread of contamination in the ROK, ensuring the drinking water provided by wells does not pose a threat to human health and mission capability.

The Department of Defense's (DoD) ability to remediate contaminated sites overseas is hindered by the availability of money. While continental US (CONUS) based remediation efforts are financed by Congress through the Defense Environmental Restoration Program (DERP), the use of DERP funds is prohibited at overseas bases

(Griffin, 1998). The remediation of contaminated sites overseas must be paid for using the operation and maintenance (O&M) funds for that base or command and must compete with other mission requirements and installation priorities. Thus, particularly at overseas bases, there is an urgent need for inexpensive and effective groundwater remediation technologies.

1.2 Background

There are two basic strategies for managing contaminated groundwater: removing the contaminant source or containing the plume generated by the source (Mackay and Cherry, 1989). A chlorinated solvent, when spilled, moves as a separate phase liquid. Since these solvents are heavier than water, they are known as dense non-aqueous phase liquids or DNAPLs. DNAPLs often sink to the bottom of aquifers, leaving residual contaminant "blobs" as they travel through the aquifer, and pooling atop low permeability formations. A DNAPL contamination plume source area is very difficult to cleanup (or even locate, for that matter). Since there is such difficulty in removing the source, remediation typically depends upon containment of the groundwater "plume" of dissolved contaminant (Mackay and Cherry, 1989). Containment involves intercepting the plume and treating the water in a treatment system to levels that do not pose a human or environmental hazard.

There are three commonly used methods for containing groundwater plumes of chlorinated solvents: pump-and-treat, permeable reactive barriers (PRBs), and natural attenuation. Pump-and-treat systems pump water from contaminated aquifers to aboveground treatment facilities for remediation. The treated effluent can then be used, discharged to surface water, or returned to the aquifer (Stoppel, 2001). An advantage of the system is its ability to actively capture a contaminated groundwater plume that may be located at a considerable depth below the ground surface. However, disadvantages are the costs and risks of bringing contaminated water to the surface.

PRBs are often configured as a "gate" in a funnel-and-gate configuration. The PRB gate is filled with a reactive media (typically consisting of zero-valent iron and sand or gravel). Contaminated groundwater is directed towards the gate through strategically located piles (the "funnel"). As the groundwater contaminated with a chlorinated solvent passes through the gate, the contaminant is chemically or biologically destroyed. An advantage of the funnel-and-gate system is that the remediation is conducted *in situ* (that is, in place in the subsurface), avoiding collection and disposal problems. However, clogging can occur in the PRB due to precipitation or biological growth, and since the technology is passive (there is no active pumping) the possibility exists for the contaminant plume to bypass the PRB (Stoppel, 2001). In addition, the depth to which the PRB can be installed is typically limited to a depth not exceeding 10 meters (Vidic and Pohland, 1996), so the technology is not applicable for deep groundwater plumes.

Natural attenuation relies upon physical, chemical, and biological processes occurring in the subsurface to destroy the contaminant. The EPA defines monitored natural attenuation as:

> …the reliance on natural attenuation processes (within the context of a carefully controlled and monitored clean-up approach) to achieve site-specific remedial objectives within a time frame that is reasonable…[They] include a variety of physical, chemical, or biological processes that …act without human intervention to reduce mass, toxicity, mobility, volume, or concentration of contaminants in soil and groundwater. (EPA, 1997)

While natural attenuation may be effective if the appropriate subsurface conditions exist, the process may take too long to be protective of human health and the environment (Young, 2001). In addition, the hydrologic and geochemical conditions favoring significant biodegradation of chlorinated solvents sufficient to achieve remediation objectives within reasonable timeframes are only anticipated to occur in limited circumstances (EPA, 1997). Additionally, some geologic formations such as fractured bedrock aquifers or limestone areas are not likely candidates for natural attenuation due to unpredictable ground water flow and difficulty in predicting the movement of contaminants (EPA, 1996). Other disadvantages include costly site characterization and negative public views, since relying on natural attenuation is often seen as a do-nothing approach (EPA, 1998a).

Horizontal flow treatment wells (HFTWs) are an innovative technology that is being considered for use in containing groundwater contamination. HFTWs rely upon pairs of dual-screened treatment wells, with one well pumping in an up flow mode and the other in a down flow mode. Since they are dual-screened wells, the water will circulate between the wells (Figure 1.1). With each pass of water through a well, the contaminated groundwater is treated to a certain extent, determined by the in-well treatment technology. Due to the recirculation between wells, contaminated groundwater is treated multiple times, so overall contaminant treatment (comparing contaminant concentrations upgradient and downgradient of the HFTW system) is greater than treatment achieved by a single pass of contaminated water through a single treatment well.

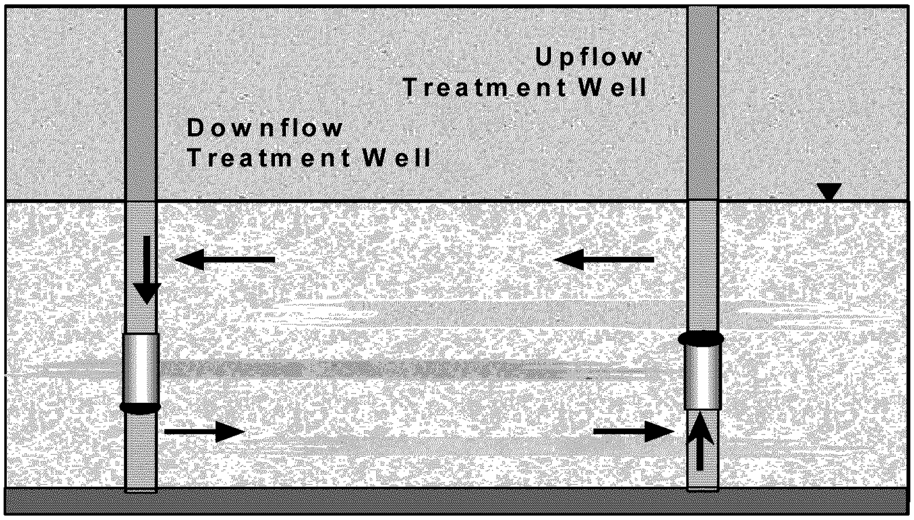


Figure 1.1: Schematic of HFTW System

An HFTW system achieves active containment with *in situ* treatment, thereby combining the best characteristics of both the pump-and-treat and funnel-and-gate systems. Additional advantages of the HFTW system are:

> …(1) the costs and risks of pumping contaminated groundwater to the surface are avoided, (2) no aboveground treatment system is required, (3) the contaminant is destroyed and not simply concentrated in another medium for disposal, (4) disposal of treated groundwater is not an issue, and (5) uncontaminated groundwater is not wasted by being brought into the contaminated zone as generally occurs in pump-and-treat systems. (McCarty, 1998)

HFTWs that incorporate either in-well catalysts or bioreactive zones have been proposed to contain chlorinated ethene contaminated groundwater (Christ, 1997; Stoppel, 2001; Christ *et al.*, 1999; McCarty *et al.*, 1998). HFTWs have already shown the potential, in CONUS evaluations, to be a cheaper, safer, and more effective technology for the remediation of chlorinated solvent contaminated groundwater, with removal efficiencies of 97 – 98% (McCarty *et al.*, 1998; Christ, 1997). Since the remediation is performed *in situ*, HFTWs have small aboveground footprints making them attractive for use in the ROK due to limited land availability. Thus, this study will focus on the remediation of groundwater contaminated with chlorinated solvents at DoD installations in the ROK utilizing HFTWs as a cheaper, safer, less obtrusive, and more effective technology.

1.3 Research Objective

Remediation in the ROK must meet space and funding constraints while being safe and effective. The goal of this research will be to examine conditions at USFK installations in Korea to see if HFTWs might be an appropriate remediation technology to deploy there. In order to accomplish this, the research will focus on the following questions:

1. Under what site conditions does HFTW technology out-perform other technologies (considering operating and installation costs, efficiency, and safety)?
2. What are the characteristic site conditions at USFK installations in Korea?
3. Assuming HFTWs are appropriate, how can HFTW technology be applied at USFK installations in Korea?

1.4 Scope and Limitations

The research is limited to the evaluation of HFTW applicability in the ROK for the treatment of chlorinated solvent contaminated groundwater. The data produced by this study will provide remediation program managers with information on HFTW performance under hydrogeologic conditions encountered in the ROK.

Specific limitations are as follows:

1. Analysis of HFTW performance is based on a single field study of HFTWs that was conducted at Edwards Air Force Base in California (McCarty *et al.*, 1998).

2. Contaminant and hydrogeologic data from US installations in the ROK are somewhat sparse, for the reasons discussed in Section 1.1. Data that are available, from either US or ROK sources, will be used to make generalizations regarding contaminant hydrogeology at USFK installations.

2 Literature Review

2.1 Introduction

In this chapter, we review literature related to the remediation of chlorinated solvent contaminated groundwater in the Republic of Korea (ROK). In the first section, we review conditions in the ROK relevant to remediation of subsurface contamination at US installations. In the second section, we examine technologies appropriate for the containment and treatment of chlorinated solvent contaminated groundwater, with an emphasis on how the technologies might be applied under the conditions found in the ROK. In the third section, we present case studies where technologies were used to treat chlorinated solvent contaminated groundwater under hydrogeologic conditions similar to those encountered in the ROK and present modeling techniques that may be used to simulate contaminant fate and transport under those same hydrogeologic conditions.

2.2 Conditions in Korea relevant to remediation of subsurface contamination at US installations

In this section, we review both the "social" (e.g. political, regulatory) and physical conditions that affect management and remediation of subsurface contamination at United States Forces Korea (USFK) bases located in the ROK.

2.2.1 Regulatory/Policy

Current policies regarding contaminant remediation at US facilities in the ROK are primarily the result of Article IV of the Mutual Defense Treaty between the Republic of Korea and the United States of America, regarding Facilities and the Status of the United States Armed Forces in the Republic of Korea (henceforth referred to as the Status of Forces Agreement or SOFA). Article IV of the SOFA states:

> The government of the United States is not obliged, when it returns facilities and areas to the Government of the Republic of Korea on the expiration of this Agreement or at an earlier date, to restore the facilities and areas to the condition in which they were at the time they became available to the United States armed forces, or to compensate the Government of the Republic of Korea in lieu of such restoration. (USAF, 1981)

It is important to note that the current SOFA explicitly states that the US will not be monetarily liable for remediation costs. Further guidance can be found in Department of Defense Instruction 4715.5, Management of Environmental Compliance at Overseas Installations. This instruction states that funds for environmental remediation will be allocated only if leaving spill sites unremediated will pose "(a)n imminent and substantial threat to human health." (DoD, 1996) Air Force Instruction (AFI) 32-7006, Environmental Program in Foreign Countries, discusses the four environmental pillars: cleanup, compliance, conservation and pollution prevention. It states that the Air Force is responsible for executing cleanup projects to the point that contamination no longer poses an imminent and substantial danger to human health and safety and as needed to sustain current operations unless the Air Force is bound by international agreement to do more (USAF, 1994).

While the current mutual defense treaty and Status of Forces Agreement (SOFA) between the US and ROK governments do not require remediation of environmental contamination at USFK installations, increasing pressure to change US remediation policy was seen in recent negotiations which amended the SOFA in part to address environmental pollution around USFK bases. The newly agreed to SOFA required that the US and ROK sign a memorandum of special understanding to include cooperative measures for environmental protection (Lea, 2001). Under the new memorandum, the United States is required to: (1) review and update Environmental Governing Standards biennially for the purpose of accommodating more protective rules and standards; (2) work together and exchange information that could effect the health and environment of the Republic of Korea and USFK members; (3) consult with the ROK on any risks posed by environmental contamination on USFK facilities and areas, or in communities adjacent to such facilities and areas by conducting periodic environmental performance assessments that examine, identify and evaluate USFK operations in order to minimize adverse environmental effects; (4) plan, program, and budget for these environmental requirements accordingly; and (5) promptly undertake actions to remedy contamination caused by United States Armed Forced in Korea that poses a known, imminent and substantial endangerment to human health (DoD, 2001). Robert T. Mounts, the US SOFA secretary in Seoul, stated:

> We now have an agreed minute that says both the United States and South Korean governments recognize the importance of environmental protection, that we commit to implement the SOFA consistent with protection of the environment and health and we will conform our policy to respect South Korean environmental laws (Lea, 2001).

The SOFA and memorandum of special understanding negotiations came after precedents were set at US military installations in Germany, Canada, and Panama, where the US either funded remediation efforts or paid substantial penalties for turning contaminated installations back to the host nations (Oshiba, 1997; Shin, 2001). Even more recently, on 17 May 2001, the House of Representatives passed an amendment to the Foreign Relations Authorization Act for Fiscal Years 2002 and 2003. The amendment supports an assessment to examine environmental contamination and health effects emanating from former US military facilities in the Philippines (Brooks, 2001; Gault, 2001). While not directly related to US activities in Korea, the amendment, taken in conjunction with the precedents set in Germany, Canada, and Panama, indicates that there is the possibility that the US will accept responsibility for remediation of contamination at DoD installations in Korea sometime in the future. A more in depth review of US Department of Defense environmental remediation policies in the ROK can be found in Oshiba's (1997) thesis.

2.2.2 Political

The Korean public has become increasingly aware of environmental conditions at USFK installations. In fact, one result of this increasing awareness was the revised SOFA discussed above. Two recent incidents served to demonstrate the sensitivity of the Korean public toward environmental contamination at USFK installations. The first incident involved the dumping of formaldehyde into the Han River in Seoul via the storm sewer (Kim, 2001; Shin, 2001; Son, 2001). After the source of the formaldehyde was discovered to be the US 8th Army Mortuary, civic groups protested and delivered a letter to the Korean government requesting a case be opened against the deputy chief of the mortuary (Lee, 2001; Son, 2001). The Korean government opened a case, and is pushing for jurisdiction to try the deputy chief of the mortuary under an indictment by a summary court. Korean Trade Ministry officials state that Article 22 of the Agreed Minutes of the ROK-US SOFA, which covers criminal acts by civilians, grants Korean courts jurisdiction (Son, 2001). United States Forces Korea (USFK) responded to the charges with an official apology from the commander of the 8th US Army, and an explanation that the chemical posed no hazard to the environment if diluted in water. The second incident involved a fuel spill that polluted over 6,000 square meters of farmland at Camp Long (Lee, 2001). As in the Han River case discussed above, civic leaders again filed a complaint with the government asking that negligent USFK personnel be indicted (Lee, 2001).

In addition to the increased public awareness, as illustrated above, some environmental groups are taking action, like conducting fact-finding surveys to determine the extent of pollution surrounding USFK bases. In the Kyonggi province, the provincial government recently teamed with an environmental group to survey pollution at 14 US bases in the province (Kang, 2001). The survey will look at all types of pollution, including oil, noise, and the dumping of wastewater and solids. Provincial leaders are hopeful that this survey will result in movement toward a stricter environmental policy than what is currently in the SOFA, which they claim lacks concrete steps for (1) punishing USFK members responsible for polluting the environment and (2) collecting compensation for environmental damage (Kang, 2001).

2.2.3 Contaminant hydrogeology

Site data obtained by Hartman (1999) and Lee (1999) indicate relatively uniform hydrogeologic patterns in the ROK. Site geology can be characterized as (sequentially from the surface) alluvium and/or reclaimed soil, 5-30' thick, and bedrock consisting of biotite schist, gneiss, or granite (Lee, 1999; Hartman, 1999). Groundwater depths typically range from 2-15' below ground surface. Corings and well drilling logs at Camp Long and Osan AB show that the groundwater flows through fractured media that can be water bearing up to 900 feet below the overburden. A representative well drilling log is presented in Figure 2.1 from a contaminated well at Osan AB. Additional drilling logs from Osan AB that were used to characterize the hydrogeology of the site can be found in Appendix 1.

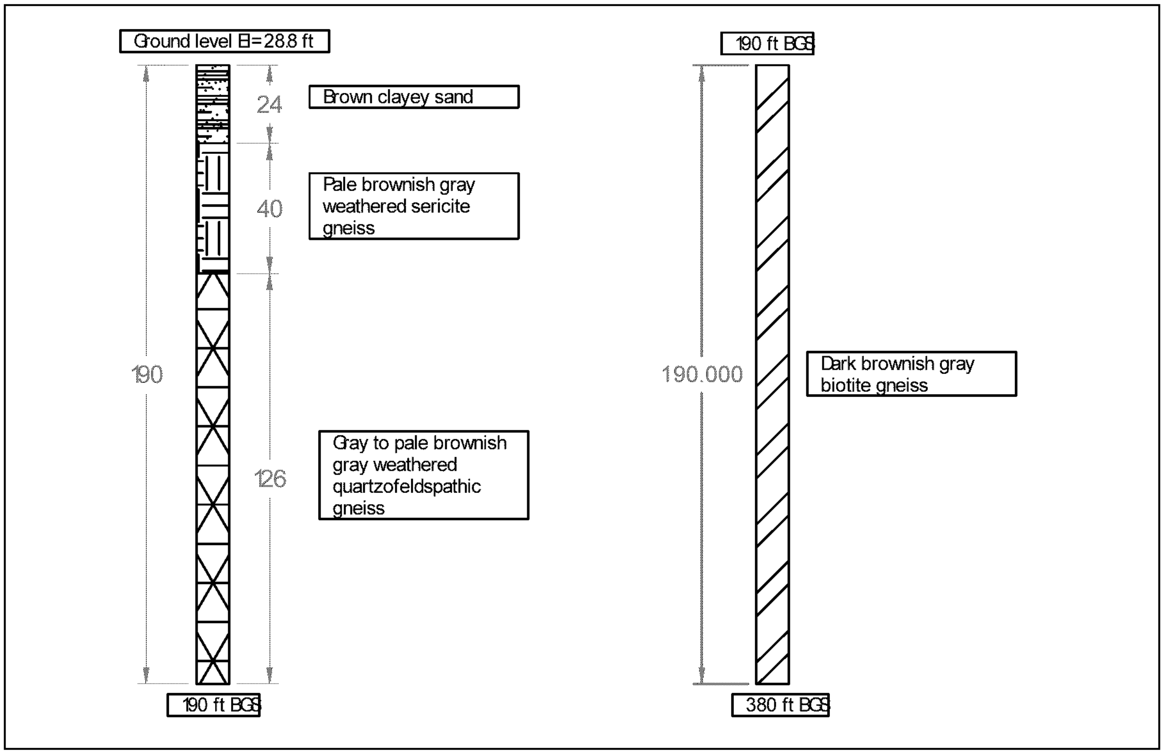


Figure 2.1: Soil data from well at Osan AB, ROK

Fractured media systems behave differently than porous media, in that the water flow is generally attributed to discrete fractures. Fractures in rock aquifers can be caused by: 1) response to faulting or folding, 2) deep erosion of overburden that can cause differential stress, or 3) rock volume shrinkage or expansion due to temperature differentials and water loss or gain (Domenico and Schwartz, 1998). Weathering can increase bedrock fracture density three to four orders of magnitude over unweathered tills (Domenico and Schwartz, 1998). The well boring logs at Osan AB (see Appendix 1) support that the bedrock is weathered to differing depths.

Information gathered by Hartman (1999) and information provided by the Army Corps of Engineers Far East District identify some of the contaminants located at USFK installations. While the site characterization of all USFK bases for contaminant type, amount, and plume extent is incomplete, the available information is believed to be representative of the contaminant types at various USFK installations. The predominant contaminant present was fuel hydrocarbons, though chlorinated solvent contamination appeared as well, particularly at the two Air Force main operating bases, Osan and Kunsan. At Osan, there is evidence of chlorinated solvents penetrating deep within the fractured media, making source removal impracticable (NRC, 1994; Osan AB CE, 2001). Nine locations were characterized as being chlorinated solvent sources at the two bases combined, indicating a potentially widespread problem at all USFK bases (Hartman, 1999; Osan AB CE, 2001).

Characteristic well depths at Osan AB range from 200' to 400' (Figure 2.2). The upper portions of the wells that run through the overburden (50'-60' in depth) are typically cased. The lower portions drilled into the fractured bedrock are not screened, but rather left open to the bedrock. Samples taken from these wells contain TCE at concentrations of 83 μg/L and vinyl chloride at concentrations of 20 μg/L (Osan AB CE, 2001). Note, however, that these sample concentrations are essentially vertically averaged over the uncased depths of the wells. When the wells are pumping, dynamic water levels that are 100' to 200' below ground surface have been measured. Thus, the detection of contaminants in water samples from the wells indicates that contamination is relatively deep (below the dynamic water level). Due to mixing in the well bore, it is also likely that contamination levels at certain depths are much higher than the concentrations measured in the samples, which, as noted above, have been vertically averaged. These high concentrations are probably found in permeable zones (where the fractures are numerous and connected), where water flow and contaminant transport is relatively rapid (Figure 2.2).

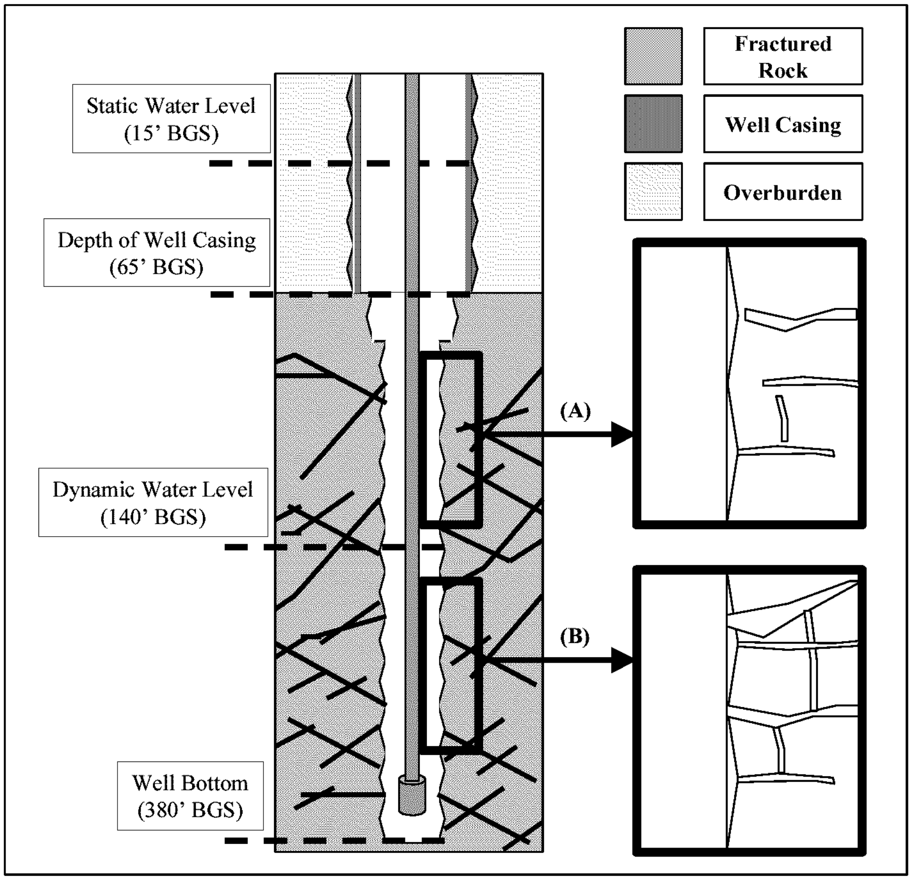


Figure 2.2: Well construction detail showing: (a) low permeability zone and (b) high permeability zone (numerous fractures with high connectivity)

When an organic contaminant such as TCE is spilled, pure phase chemical (known as non-aqueous phase liquid or NAPL) is transported through the vadose zone to the water table. In a typical spill, most of the contaminant is in the NAPL phase and chemical in the dissolved phase represents only a small fraction of the total contaminant present. Thus, one of the primary challenges of groundwater remediation is the removal of the separate phase NAPL that serves as a subsurface source, which allows plumes of dissolved contaminant to grow and persist. When the source cannot be removed effectively, containment of the dissolved plume becomes the goal. Particularly for dense NAPLs (DNAPLs) like TCE and other chlorinated solvents, which due to the fact that they have a specific gravity greater than unity sink below the water table, there has been little success locating or removing the sources. Thus, remediation for aquifers contaminated by DNAPLs often relies upon containing the dissolved plume, rather than removing the source (Mackay and Cherry, 1989).

When NAPL enters fractured systems, it flows mainly through interconnected fractures and settles out in dead-end segments of the fracture system. NAPL can move deep and far into the rock, entering dead-end fractures and eventually becoming immobile. The prognosis for cleanup of fractured rock aquifers, particularly those containing NAPLs, is worse than for sand and gravel aquifers because of the contaminant immobility, leaving containment of the plume as the only viable management option (Mackay and Cherry, 1989).

2.3 Chlorinated solvent contaminant plume remediation technologies

The goal of groundwater remediation is to reduce contamination in an aquifer to levels that are protective of human health and the environment at the lowest cost and in an expeditious manner (Bumb, 1997). In order to accomplish this goal, remediation practices should focus on removing the contaminant mass that serves as a subsurface source, which causes plumes to grow and persist, rather than simply removing the mass of dissolved contaminants that defines the plume (Mackay and Cherry, 1989). However, as noted above, data collected from bases in the ROK show that spilled chlorinated solvents have penetrated through the overburden and are present deep within the fractured media, making source removal difficult if not impossible. Therefore, this thesis will concentrate on containment of the plume versus source removal. A plume containment system should: (1) capture and treat the contaminated ground water plume, (2) not increase the areal extent of soil contamination by inducing flow of contaminants through uncontaminated aquifer media, and (3) minimize the aboveground "footprint" of the system (Bumb, 1997).

This section reviews the most common strategies and technologies that are employed to contain plumes of groundwater contamination. In addition to explaining the principles of operation, this section will review the application of these technologies and strategies in fractured media systems.

2.3.1 Pump-and-treat

Pump-and-treat systems are based on a relatively simple process. Wells are placed into an aquifer and contaminated water is pumped to the surface where it is subsequently treated. After treatment, the water can be discharged into surface water bodies, re-injected into the aquifer, or percolated back into the aquifer using recharge trenches (Mackay and Cherry, 1989; NRC, 1994). As a result of pumping contaminated water, clean water is drawn into the site, often reducing the contaminant concentrations. A leveling of concentration, and a gradual decline that can take decades, follows the initial contaminant reduction (Mackay and Cherry, 1989; Cartwright, 1991). Pump-and-treat systems can be used to either control contaminant migration, or to remove the contaminant source. Since source removal in the fractured aquifer systems found in the ROK is infeasible, containment is the goal. Thus, a pump-and-treat system should be designed and installed to minimize pumping while controlling the spreading of the plume (NRC, 1994).

2.3.1.1 Advantages

Pump-and-treat technology has been in widespread use since 1985 (NRC, 1994) and dominates the field of contaminated aquifer remediation (Cartwright, 1991). As such, it is the most thoroughly researched and documented technology, widely accepted by regulators and the groundwater remediation community (Shanley, 1996; Roote *et al*., 1997). It has been applied with limited success in fractured media aquifers in the past as a means of containment (Gaule *et al*., 1993; Mackay and Cherry, 1989; NRC, 1994). Case studies for containment in fractured systems using pump-and-treat are presented in section 2.4.

The principal advantage of pump-and-treat is that it is an active system. Pumping controls plume migration, minimizing the chance of the contaminant bypassing the system due to changing hydrogeologic conditions (Stoppel, 2001). In addition, standard hydrogeologic and engineering practices make the systems relatively easy to design (Roote *et al*., 1997).

2.3.1.2 Disadvantages

The ubiquity of pump-and-treat remediation systems has encouraged numerous reviews of their performance. Pump-and-treat wells, through active pumping, can lower the concentration of contaminants due to mixing of contaminated water with uncontaminated water. This contamination of previously uncontaminated water is a disadvantage of pump-and-treat (McCarty *et al*., 1998). An additional disadvantage of pump-and-treat systems is the requirement to pump contaminants to the surface for collection or destruction (Christ, 1999; Ferland, 2000;). The long-term operation of a pump-and-treat system is often expensive due to the energy used to pump and treat large volumes of water and effluent disposal costs (Mackay and Cherry, 1989; Roote *et al*., 1997).

Pump-and-treat, although it has been applied in the past, has not been entirely successful in the containment of chlorinated solvents in fractured media aquifers and may not be an appropriate strategy (Gaule *et al*., 1993; Mackay and Cherry, 1989; NRC, 1994). Case studies presented in section 2.4 indicate contaminant transport occurs slowly due to limited solubility, and large volumes of water must be treated in order to achieve containment for the duration.

2.3.2 Natural attenuation

Natural attenuation relies on naturally occurring physical, chemical, and biological processes to reduce the risk posed by groundwater contamination to acceptable levels (EPA, 1996). Natural attenuation can destroy contaminant mass through biodegradation and chemical transformations, or non-destructively remove contaminants through dilution, dispersion, and adsorption (EPA, 1996). In studies involving chlorinated solvents, microbially mediated dehalogenation occurred without intervention, potentially protecting human health and the environment (Stoppel, 2001). Implementation of natural attenuation as a remedy at a site, according to Murray *et al.* (1999), should be accompanied by assurances that the process will protect human health and the environment and achieve remediation goals within a reasonable time frame.

2.3.2.1 Advantages

The EPA (1996) cites several reasons why natural attenuation may be advantageous. The primary reason is that it can be effective and inexpensive. Although sometimes labeled as a "do nothing approach," natural attenuation is in reality a proactive process of confirming and monitoring the natural elimination of contaminants. Under proper site conditions, *in situ* remediation occurs, allowing the use of land above the contaminant plume. Additionally, because the contaminant is being treated non-invasively, no construction or mechanical system costs are incurred, and no energy source is required for pumping water (EPA, 1996).

2.3.2.2 Disadvantages

In addition to sometimes being seen as a "do nothing approach" by the public, natural attenuation has several limitations and disadvantages that are a result of contaminant hydrogeology. The principal disadvantage is the long amount of time it may take natural processes to rid an aquifer of a contaminant (Ferland, 2000; Stoppel, 2001; Murray *et al*., 1999). The Office of Solid Waste and Emergency Response (OSWER) states that "the hydrogeologic and geochemical conditions favoring significant biodegradation of chlorinated solvents sufficient to achieve remediation objectives within a reasonable timeframe are anticipated to occur only in limited circumstances" (OSWER Directive 9200.4-17P, 1999). Also, in the case of chlorinated solvent contamination, natural attenuation can sometimes create daughter products that are more harmful than the primary contaminant (Wiedemeir *et al*., 1998).

Determining if natural attenuation is appropriate for a site and monitoring the contamination to assure natural attenuation is operating can incur significant costs, and the results of such investigations may yield inconclusive results (EPA, 1997; Wiedemeir *et al*., 1998).

Based on the above, it appears that natural attenuation is not an appropriate strategy to apply to chlorinated solvent contamination in fractured bedrock aquifers. Even under the best conditions, natural attenuation of chlorinated solvents is problematic. In a fractured aquifer, with unpredictable groundwater flow and difficult and expensive monitoring conditions, using natural attenuation as a chlorinated solvent remediation strategy would most likely be infeasible.

2.3.3 Permeable reactive barriers

One method for containing a contaminant plume is through the use of a permeable reactive barrier (PRB). A PRB is a permanent, semi-permanent, or replaceable unit placed across the flow path of a contaminant plume. Passive *in situ* treatment occurs when reactive material degrades or immobilizes contaminants as groundwater flows through it. Natural gradients transport the contaminants through strategically placed treatment media (EPA, 1999). PRBs can be installed so as to intercept the entire width of the plume, or the amount of reactive media can be minimized through utilizing a system known as a funnel-and-gate. With a funnel-and-gate, impermeable slurry walls or sheet piles are installed to funnel the contaminated groundwater flow into a PRB (the gate). A hanging gate that does not penetrate all the way to a lower confining layer can be used for shallow plume containment (for example, for containing a plume from an LNAPL source), while a fully penetrating gate may be needed for containment of a deep plume, as is shown in Figure 2.3.

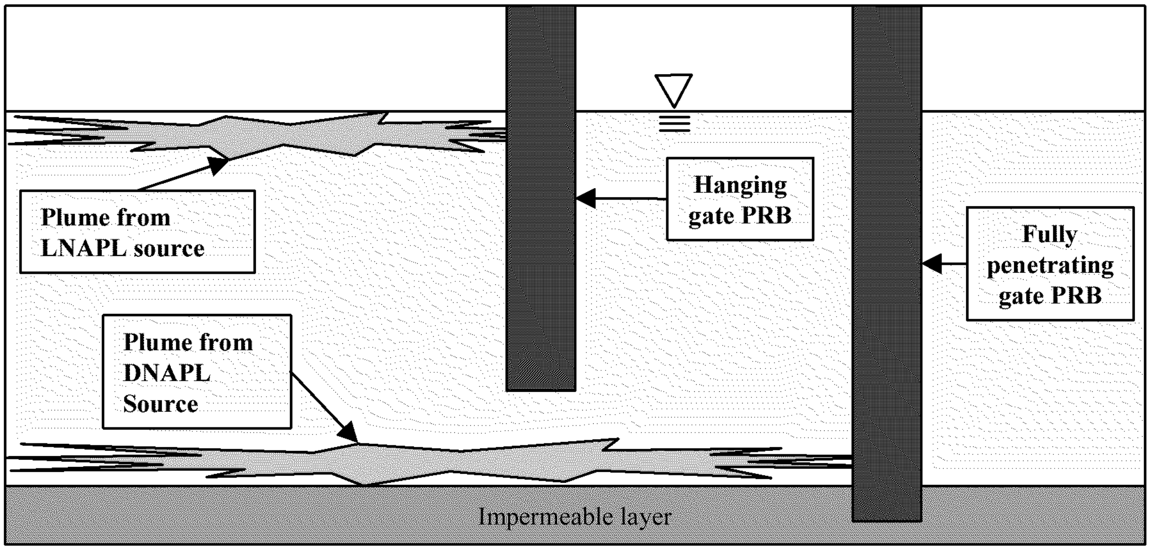


Figure 2.3: Fully penetrating and hanging gate PRB configurations

The type of PRB determines how containment is achieved. Contaminant containment or destruction in the PRB can occur through: volatilization, microbial degradation, adsorption, chemical oxidation, metal-enhanced dechlorination, and metal precipitation. The type of contaminants in the groundwater will determine the amount and types of PRBs necessary (Starr and Cherry, 1994; Grindstaff, 1998; Blowes, 2000; Sedivy, 1999). Blowes *et al.*, (2000) reported that the costs associated with the design, installation, site rehabilitation, and monitoring of a funnel-and-gate or PRB should provide treatment at costs comparable to other treatment methods.

2.3.3.1 Advantages

PRB technologies have many of the same advantages of natural attenuation since *in situ* remediation is occurring with little or no impact on the surface, minimal operating and maintenance costs during operation, and no energy costs due to pumping. PRBs are advantageous over natural attenuation since the water is attenuated while flowing through the barrier, greatly reducing treatment time. The funnel-and-gate technology enables rapid construction, and a reduced volume of reactive media required as compared to a PRB that intercepts the entire plume width (Roote *et al*., 1997; Sedivy *et al*., 1999).

2.3.3.2 Disadvantages

While the PRB system has many advantages, it unfortunately is useful for only a narrow range of hydrologic conditions. The installation depth is limited due to cost and construction constraints, and typically does not exceed 10 meters (Vidic and Pohland, 1996; Stoppel, 2001). Variations in groundwater flow over time can also allow the contaminant to bypass the treatment system (McMahon *et al*., 1999; Ferland, 2000; Stoppel, 2001). There are several conditions which may arise within the PRB that could impact the efficiency of the system: (1) microbial and precipitate clogging that could decrease porosity (Grindstaff, 1998; Blowes *et al*., 2000), (2) preferential channeling of water through the PRB could deplete reactants in those zones more rapidly, and (3) the method by which the contaminant is removed may require eventual removal of contaminants from the PRB media or replacement of the PRB (Blowes *et al*., 2000). A further disadvantage is the fact that once a PRB is emplaced, the technology can only be modified with great difficulty and expense. If the contaminant concentration or the flow of groundwater into the barrier exceeds design parameters, the only remedy is to modify the barrier, unlike a pump-and-treat system, where the pumping rate could be adjusted. Starr and Cherry (1994), however, point out that installing multiple or sequential treatment barriers could remedy this.

Due to depth limits of PRB technology, it seems unlikely that it would be practical for conditions in Korea, where contaminant has been found at considerable depth.

2.3.4 HFTWs

Unlike the technologies previously discussed, which are in common use, HFTWs are an innovative technology that has only been tested at Edwards AFB in California. HFTWs consist of two or more dual screened wells alternately pumping in upflow or downflow modes. In a two-well system, one well will operate in the upflow mode lifting water from the lower portion of an aquifer into the upper portion, while the other well will pump in a downflow mode, injecting water from the upper portion of the aquifer into the lower portion, as is shown in Figure 2.4. Aquifer anisotropy is critical for an HFTW system to properly work, as there should be minimal vertical flow of groundwater between the injection and extraction screens of a single treatment well (Stoppel, 2001). This is typically found to be the case, since most aquifers exhibit horizontal hydraulic conductivities an order of magnitude greater than vertical hydraulic conductivities (Domenico and Schwartz, 1998). Fractured rock typically has an anisotropy up to two orders of magnitude (Domenico and Schwartz, 1998; Charbeneau, 2000).

Two techniques for employing HFTWs are depicted in Figures 2.4 and 2.5. In Figure 2.4, in-well reactors treat contaminated groundwater (Ferland, 2000). In Figure 2.5, nutrients are added in-well to promote biodegradation by bacteria in bioactive zones outside the injection screens of the treatment wells (McCarty *et al.*, 1998).

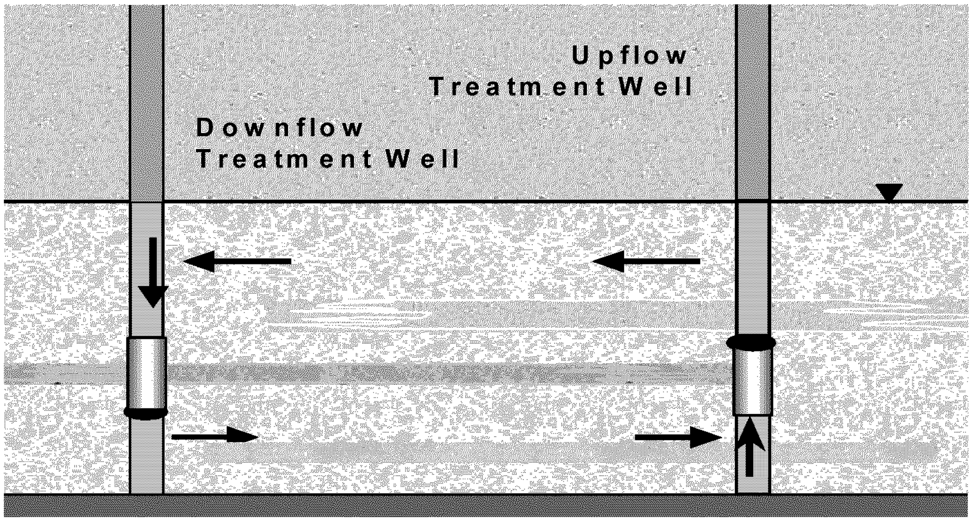


Figure 2.4: Schematic of HFTW system with in-well Pd-catalyst reactor

Contaminant concentrations downgradient of an HFTW system (as compared to upgradient concentrations) depend upon two factors, contaminant destruction during a single-pass of contaminated water through the treatment well (defined as the single-pass treatment efficiency) and the fraction of treated water that is recycled through the treatment wells (Christ *et al.*, 1999). The single-pass treatment efficiency depends upon the treatment method and the residence time of the contaminant in the reactor or bioactive zone. The fraction of water recycled through the system depends upon the hydrogeologic conditions at the treatment site, the flow rate in the wells, the distance between wells, etc. To apply this technology, a line of treatment wells, as shown in Figure 2.6, may be located downgradient of a contaminant plume, actively containing the plume (Ferland, 2000).

2.3.4.1 Advantages

HFTWs possess advantages intrinsic to both pump-and-treat and PRB remediation strategies. HFTWs actively control plume migration like pump-and-treat, reducing the chance of the contaminant bypassing the remediation system as might occur with PRBs. Like PRBs, the treatment is *in situ*, thereby reducing the costs and health risks associated with pumping contaminated water to the surface (Christ *et al*., 1999; Stoppel, 2001). Additional advantages of subsurface contaminant destruction are relief from regulatory burdens, as there is no requirement to permit aboveground treatment facilities or for disposal of treated water, and the footprint of the treatment facility is limited as the reactor can be installed beneath the ground surface (McCarty *et al*., 1998; Christ *et al*., 1999; Lowry and Reinhard, 2000;). As no water is extracted, uncontaminated groundwater is not wasted by being brought into the contaminated zone, as occurs in pump-and-treat systems. For this reason, HFTWs are especially advantageous in areas that experience water shortages (McCarty *et al*., 1998). The active nature of the HFTW system also provides greater flexibility and hydraulic control than PRBs, allowing contaminant capture at significant depths under fluctuating flow conditions (Ferland, 2000).

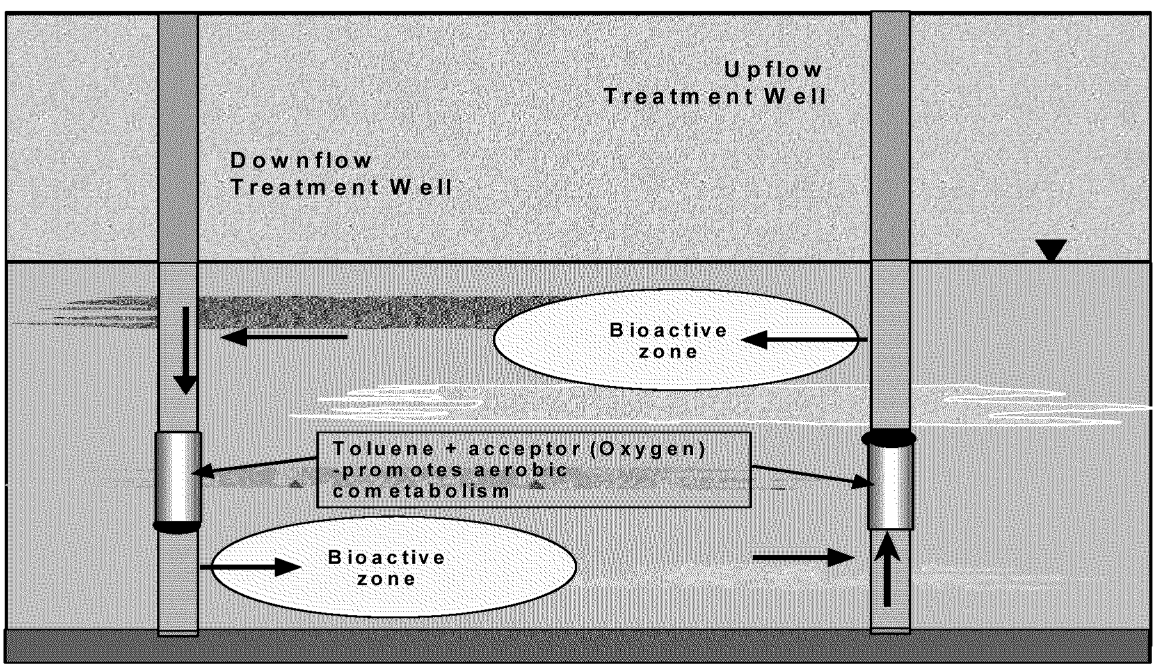


Figure 2.5: Schematic of HFTW system with bioactive zones

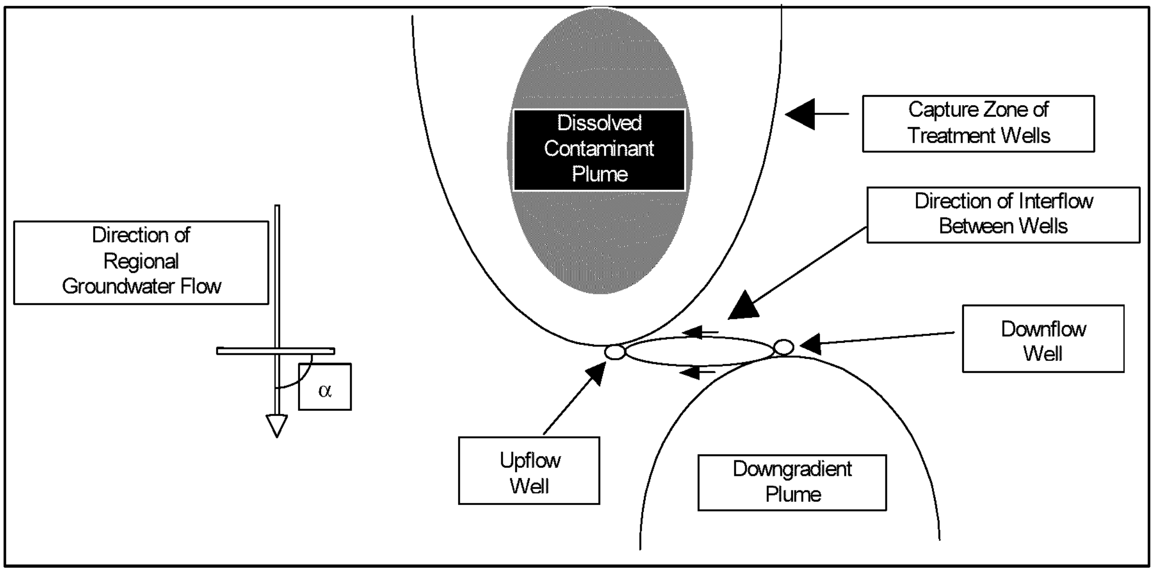


Figure 2.6: Plan view for in situ treatment by HFTWs (depicting the lower portion of the aquifer, where the downflow well is an injection well and the upflow well an extraction well)

2.3.4.2 Disadvantages

Regardless of the type of HFTW (chemical or biological), the technology is new and relatively untested. With the exception of the data from the Edwards AFB demonstration (McCarty *et al*., 1998), it is unknown how they will perform in differing site conditions. As discussed earlier, the aquifers in the ROK are predominantly fractured bedrock, and the capabilities of HFTWs to effect chlorinated solvent containment in a fractured system are unknown. Recurring or emergency maintenance could also be a problem, since the treatment equipment associated with HFTWs is located within the well-bore, and is not easily accessible.

One of the main obstacles to the application of *in situ* HFTW biological remediation is the transport of nutrients and substrate to microbes. Therefore, as with most treatment systems, a detailed site characterization including hydrogeologic conditions, microbial activity, and aquifer conditions must be accomplished to ensure that the treatment system can clean up the contaminated site (EPA, 1998b; Grindstaff, 1998).

Utilizing biological remediation techniques can lead to the clogging of well screens and generally depends upon the aquifer characteristics (Grindstaff, 1998; McCarty *et al*., 1998). However, McCarty *et al*. (1998) discovered that hydrogen peroxide (used to deliver oxygen) lowered biomass buildup near the screens due to its bactericidal properties, but did not significantly impact the remediation process. The major operational costs associated with a biological HFTW treatment system are associated with prevention of clogging and include the cost for chemicals such as hydrogen peroxide, and well redevelopment after clogging occurs (McCarty *et al*., 1998).

HFTWs are an attractive alternative when compared to the other technologies discussed above, since their design incorporates some of the advantages, while at the same time eliminating shortfalls. As such, this thesis will explore their use for the containment of chlorinated solvent contaminated groundwater in the deep fractured rock aquifers that exist in the ROK.

2.4 Contaminant remediation by pumping in fractured systems

As discussed earlier, the ROK hydrogeology consists largely of fractured media aquifer systems. Of the technologies discussed, pump-and-treat has been employed almost exclusively in these systems, and has shown some ability to contain contaminant plumes. To illustrate the application of pump-and-treat in fractured media, we present case studies and cost data in this section. We also discuss modeling techniques that may be used to simulate contaminant transport in fractured systems. These techniques will be important in order to help design and operate HFTW systems in fractured media systems.

2.4.1 Field applications

Of the treatment strategies described above, the most widely used strategy for the treatment of contaminants in fractured systems is pump-and-treat (Gaule *et al*., 1993; EPA, 2001b). The EPA reports that out of 53 fractured systems requiring remediation, 51 were treated with pump-and-treat or a combination of pump-and-treat with another technology (EPA, 2001b). In their case study on the remediation of petroleum hydrocarbon in a fractured system, Gaule *et al*. (1993) present information regarding remediation of a fractured system using pump-and-treat. During the site characterization, the authors discovered large spatial variation in contaminant thickness, hydraulic head, and transmissivity. They concluded that the spatial variation in contaminant thickness was due to angled fractures and/or bedding planes controlling the migration of contaminant. The source of differences in head was uncertain, but transmissivity differences appeared due to the existence of fractures with widely varying permeabilities. During well pump tests, if a high permeability fracture was intercepted by the well, a high transmissivity value would result (Gaule *et al*., 1993). This variability had substantial implications with regard to design of the remediation system, since the selected remedial option, pump-and-treat, depended upon the volume of water passing through the system. The need for adequate and reliable hydrologic data to design a remediation system for fractured bedrock is necessary, and several phases of the investigation were unable to obtain needed data (Gaule *et al*., 1993). From their study, Gaule *et al*. (1993) concluded that due to the complexity of the fractured aquifer, long-term pump tests would be required to predict performance of a remediation system.

When attempts are made to remediate fractured systems by pumping water, reductions in contaminant concentrations to very low levels are rare because little or no water flushes through dead-end fracture segments or through the porous but relatively impervious rock matrix, both of which are likely to retain the bulk of the contaminated mass. Thus, a means to contain the contaminant plume must be employed. This difficulty is illustrated at a cleanup of an organic liquids disposal site in Ville Mercier Quebec, where the application of pump-and-treat technology to a large plume has been severely hampered by the penetration of NAPLs into the fractured bedrock (Mackay and Cherry, 1989).

At Ville Mercier, three pump-and-treat wells have been in operation since 1984. The site survey indicated that DNAPL pools had penetrated through the overburden and were resting on top of a low-permeability basal till directly above the fractured bedrock. In some areas, the DNAPL had penetrated into the fractures of the bedrock as a result of erratic cover by the till. The till allowed the DNAPL to continue moving downslope, but the DNAPL left behind ganglia in fractures and pores. It was determined that since the DNAPL source cannot be removed, the plume would exist for decades to centuries, and that groundwater contamination would not be eliminated from the site. Site investigators concluded that containment rather than restoration would be the goal using pump-and-treat technology (NRC, 1994). After four years and the removal of over 6 billion gallons of water, the contaminant concentrations extracted from the wells were reduced, although this reduction in concentration appeared to be due to dilution, as uncontaminated water mixed with contaminated water flowing to the wells.

Similar site conditions and results were found in King of Prussia, Pennsylvania, at a fractured bedrock aquifer contaminated with DNAPLs. Initial site investigations confirmed the presence of chlorinated compounds spilled from 1969 to 1973. The primary transport of contaminant was as a DNAPL moving along the bedding plane fractures, while dissolution created a large dissolved plume. Due to the depth and extent of the DNAPL, it was determined that the DNAPL could not be effectively recovered and that the DNAPL would continue to act as a source for dissolved contaminants in the groundwater. As a result of this site investigation, pump-and-treat technology was determined infeasible for the removal of the source of the plume, but could operate as a technique for containment of the plume (NRC, 1994).

2.4.2 Capital and Operating Costs

In this section, we review cost data for plume containment in fractured media. Cost data are provided for pump-and-treat systems, since, as indicated above, that is the only technology currently applied to contain plumes in fractured media. In addition, since an HFTW system is basically a modified pump-and-treat system, it may be possible to use the costs of pump-and-treat to estimate HFTW costs. We will also present cost data for the Edwards AFB HFTW system. Although this system was demonstrated in a porous and not a fractured medium, the data available from the Edwards demonstration are the only data available from a full-scale application of HFTWs. We therefore present the data, assuming that they may also be applicable with appropriate modifications, to a fractured media containment scenario.

2.4.2.1 Pump-and-Treat

As previously mentioned, pump-and-treat technology has been the technology of choice for plume containment. While the total cost of a pump-and-treat project is strongly dependent upon the type of aboveground treatment (vapor stripping, activated carbon, etc.), the abundance of available cost data allows quantification of a general range of project costs. Table 2.1 summarizes EPA (2001a) cost data for 18 pump-and-treat systems implemented in both fractured and porous media systems for treating chlorinated solvent contaminated groundwater. The annual operating costs were estimated based on the average volume of water treated at the 18 sites (EPA, 2001a). Included in the capital costs reported in Table 2.1 are the costs for installing the wells and the treatment system. Variations in treatment are primarily due to the type and amount of contaminant, water impurities (minerals, organics, etc.), and quantity of water being treated.

Table 2.1: Cost data for pump-and-treat systems used for treating chlorinated solvent contaminated groundwater (From EPA, 2001a)

CAPITAL COSTS ($millions)			
25th percentile	**Median**	**75th percentile**	**Average Cost**
$1.2	$1.9	$4.4	$3.6
ANNUAL OPERATING COSTS (per 1000 gallons treated)			
$3	$12	$40	$26

While Table 2.1 summarizes general costs for pump-and-treat systems, the EPA also provides cost data for a pump-and-treat system that was constructed at a contaminated fractured media site comparable to contaminated sites in the ROK. Cost figures for containing the plume at this site (King of Prussia site) are in Table 2.2. The system treated 57 million gallons of water annually (EPA, 2001a).

Table 2.2: Capital and operating costs for pump-and-treat project at King of Prussia Superfund site (From EPA, 1998c)

CAPITAL COSTS	
Equipment	$927,127
Permits	$31,637
Construction Management	$234,548
SOP/S&M Manual	$63,681
Electrical System Construction	$130,424
Other Subsystems	$194,003
Plant Construction	$131,924
Cultural Resources	$30,219
Well Construction	$116,166
Recovery System Construction	$171,707
Total Capital Costs	$2,031,436
ANNUAL OPERATING COSTS	
Labor	$108,587
Travel	$4,775
Disposal	$811
Chemicals	$16,409
Lab Supplies	$339
Health and Safety Supplies	$1,314
Administrative Expenses	$15,650
Maintenance	$53,181
Utilities	$60,500
Total Annual Operating Costs	$261,565

Note that the capital costs of the King of Prussia fractured rock system are quite close to median costs for pump-and-treat systems in porous media. The operating costs for the King of Prussia site are somewhat lower than the median costs of the other pump-and-treat systems (though still within the 25th to 75th percentile range of costs). These comparisons seem to indicate that the cost of pump-and-treat systems in porous media may be used to approximate costs for systems in fractured media.

2.4.2.2 Edwards AFB HFTW System

As HFTW technology has only been employed at one field location, limited cost data are available (Table 2.3). The system was employed in porous media, but, as noted above, it is expected that the installation cost of the wells in fractured media would be similar. The largest cost component lies in the installation of the HFTWs, while annual operating costs are minimal in comparison. Two wells capable of pumping 10 gal/min each were constructed for the Edwards AFB HFTW system (McCarty *et al*., 1998).

Table 2.3: Capital and operating costs for aerobic cometabolic *in situ* bioremediation at site 19, Edwards AFB, California (2 Wells) (From AFRL, 1998)

CAPITAL COSTS	
Treatment Wells (80 feet, 8-inch Schedule 80 PVC	$30,000
Flow Sensors and Controllers	$2,790
Static Mixers	$1,076
Packer Assembly	$9,338
Deionized Water System	$6,847
Pumps and Ancillary Equipment	$10,000
Tubing & Connectors	$1,789
Valves and Fittings	$867
Total Capital Costs	$62,707
ANNUAL OPERATING COSTS	
Well Redevelopment	$8,000
Hydrogen Peroxide, 30%	$4,633
Toluene	$47
Oxygen	$1,674
Total Annual Operating Costs	$14,354

A comparison of the pump-and-treat and HFTW systems' 20-year life cycle capital and annual costs per 1000 gallons treated adjusted for 1995 values and assuming inflation of 4% (DoD, 2000) reveals that the HFTW system is significantly cheaper (Figure 2.7). The lower annual operating costs of the HFTW system may be attributed to the *in situ* destruction of the contaminant, as well as the savings of not having to pump contaminated water to the surface for treatment.

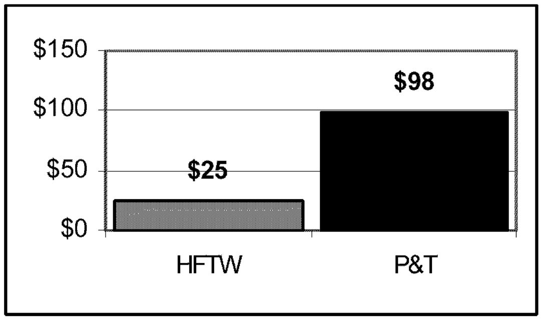


Figure 2.7: Life cycle costs (adjusted for 1995 values) per 1000 gallons treated for the HFTW system at Edwards AFB and the King of Prussia site

2.4.3 Modeling

In this section, we review the literature pertinent to modeling how the HFTW technology will perform as a contaminant containment system in a fractured media aquifer. Two key areas are discussed: HFTW modeling and modeling fractured systems. We begin with a discussion of how HFTW operation in porous media has been numerically and analytically simulated. We then investigate fractured media modeling, to determine under what conditions porous media models, such as those that have been applied to simulate HFTW operations, may be used to model HFTW operations in fractured systems.

2.4.3.1 HFTW Modeling

Modeling the efficiency of HFTWs to treat and contain contaminant plumes in porous media has been conducted in the past using both analytical and numerical models. Analytical models incorporate many simplifying assumptions. These models may be applied as a screening tool to determine if the technology is applicable at a particular contaminated site. Christ (1997) developed an analytical HFTW model for use in managing chlorinated solvent contaminated groundwater in a simple porous media. The Christ (1997) analytical model was later applied by Mandalas *et al*. (1998), Ferland *et al*.

(2000), and Stoppel (2001). Numerical models do not include many of the simplifying assumptions made by analytical models, so these models can be used to more realistically describe actual scenarios. Of course, numerical models require extensive site characterization to obtain the necessary input data. Huang and Goltz (1998) and Gandhi *et al*. (2002) have developed numerical models for application with HFTW systems. Garrett *et al*. (1999) and Fernandez (2001) used these models in conjunction with genetic algorithms to optimize HFTW application at a chlorinated solvent contaminated site.

2.4.3.1.1 Analytical Models

Christ's (1997) model made the following simplifying assumptions: (1) regional groundwater flow is steady, horizontal, and uniform in a homogenous confined aquifer of constant thickness; (2) there are an even number of pumping wells in the HFTW system, alternately injecting or extracting water at a constant rate; (3) well locations are co-linear; and (4) water flow induced by the HFTW wells is horizontal, so that the flow system into and out of the upper screens of the HFTWs can be treated independently from the flow system into and out of the lower screens. An example of a typical HFTW configuration is shown in Figure 2.8, which illustrates the flow field induced by a four-well HFTW system in the upper part of an aquifer. Note that wells 2 and 4 are upflow wells, which inject water into the upper part of the aquifer, while wells 1 and 3 are downflow wells, extracting water from the upper aquifer.

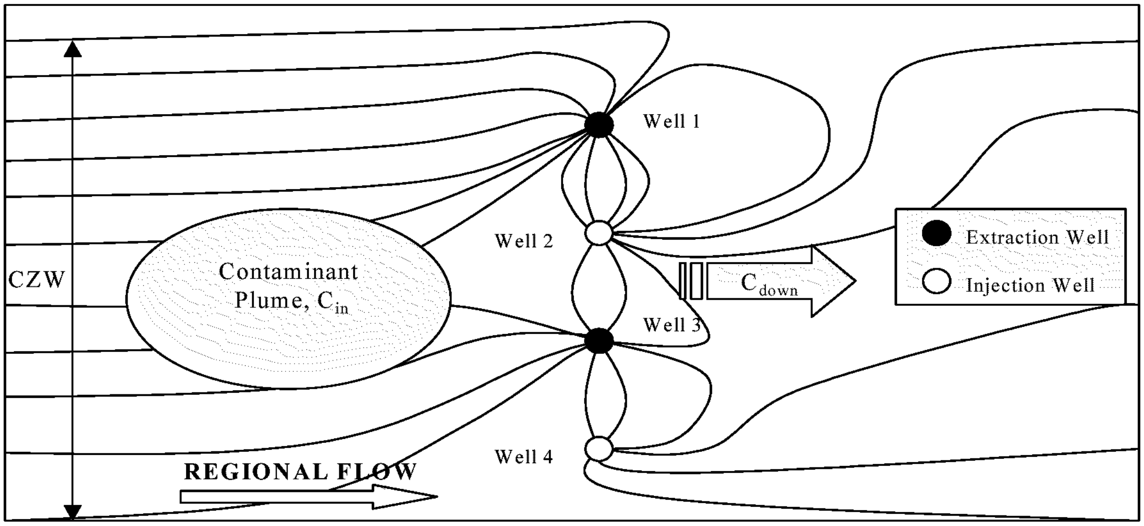


Figure 2.8: Flow field induced in the upper aquifer by four-well HFTW system (From Mandalas *et al.*, 1998)

When designing an HFTW system, the two key design variables are capture zone width (CZW) and overall treatment efficiency ($\eta_{overall}$). Capture zone width is a measure of the extent to which the contaminated groundwater plume will be captured for treatment. Overall treatment efficiency measures the extent of contaminant destruction by comparing contaminant concentrations upgradient (C_{in}) and downgradient (C_{down}) of the HFTW treatment system:

$$\eta_{overall} = 1 - \frac{C_{down}}{C_{in}}$$

Figure 2.8. illustrates these important parameters for a four-well HFTW system.

Capture zone width and overall treatment efficiency can be determined by knowing the interflow between the treatment wells in the HFTW system and the single-pass treatment efficiency of the technology being applied in the treatment wells. Interflow between two wells is defined as the fraction of the groundwater pumped through the extraction well that originated from the injection well. Christ (1997) and

Christ *et al*. (1999) present methods using complex potential theory for determining interflow based on aquifer (hydraulic gradient, hydraulic conductivity, aquifer thickness) and pumping well (pumping rate, distance between wells) characteristics. For details of these methods, the reader is referred to Christ (1997) and Christ *et al*. (1999).

The single pass treatment efficiency is defined as the fraction of contaminant destroyed following a single pass of contaminated groundwater through the treatment zone (Stoppel, 2001). Single-pass treatment efficiency is a function of the technology that is applied in the treatment wells. For an analytical model of HFTW operation, contaminant destruction is typically described as a first-order process, dependent on the residence time of the contaminant in the treatment reactor (Ferland, 2000; Stoppel, 2001). Thus, for given aquifer and well characteristics, and knowing the first-order rate constant for contaminant destruction by the technology applied in the treatment wells, a designer can analytically determine the capture zone width and overall contaminant destruction effected by an HFTW system.

Mandalas *et al*. (1998) utilized the Christ (1997) analytical model in order to develop a screening tool to help remedial project managers decide whether the HFTW technology was appropriate for application at a particular contaminated site. Ferland *et al*. (2000) and Stoppel (2001) coupled the Christ (1997) analytical flow model with simple models of single-pass treatment efficiency to estimate the overall contaminant removal efficiencies that could be obtained using an HFTW system incorporating in-well catalytic reactors under given site and operational conditions. Ferland *et al*. (2000) assumed simple first-order kinetics to estimate single-pass destruction of chlorinated ethenes in the in-well catalytic reactor. Stoppel's (2001) research incorporated catalyst deactivation and regeneration into the first-order reaction model.

2.4.3.1.2 Numerical Models

The model developed by Huang and Goltz (1998) is a three-dimensional model that combines steady-state flow, advective/dispersive transport of dissolved species, equilibrium or rate-limited sorption, and biodegradation. Huang and Goltz (1998) wrote FORTRAN code that uses a finite difference approach to numerically solve the three-dimensional partial differential equations describing fate and transport. The program MODFLOW (Harbaugh and McDonald, 1996) is used to calculate steady-state conditions of flow in the aquifer, and these flow velocities are then used in a transport model (Huang and Goltz, 1998). A finite difference grid like the one shown in Figure 2.9 is created using MODFLOW. Its dimensions and specific cell composition can be varied based on the system being modeled.

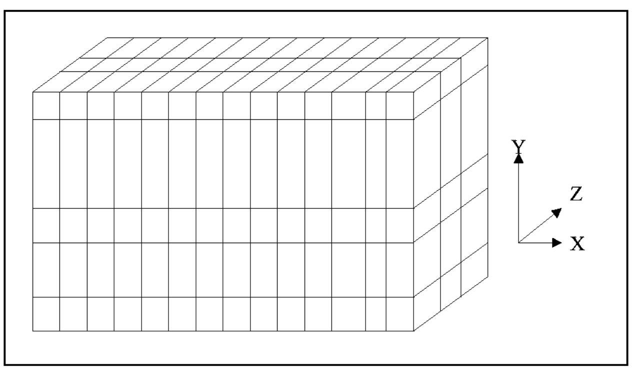


Figure 2.9: Finite difference grid used in MODFLOW (Garrett, 1999)

Well locations and associated pumping rates, initial conditions, aquifer properties (hydraulic conductivity and porosity), initial contaminant concentrations, and boundary conditions are specified within the grid. MODFLOW uses these data to calculate the steady state hydraulic head and velocity fields. The transport package of the computer program then uses the velocity data as well as concentration initial and boundary conditions to calculate concentrations over space and time. The concentrations of the dissolved species can be monitored over time at any location on the grid, which allows the user to assess simulated system performance.

Gandhi *et al.* (2002) developed a three dimensional, numerical model that was used to simulate the Edwards AFB Site 19 HFTW system. Though similar to the Huang and Goltz (1998) model, the Gandhi *et al.* (2002) model was based on finite elements. This allowed for smaller grid dimensions near wells, where high spatial resolution was needed (Gandhi *et al.*, 2002). Gandhi *et al.* (2002a) developed a flow model that described conditions at the Edwards site. For further details regarding the mathematical formulation of the site model, the reader is referred to Gandhi *et al.* (2002).

Garrett *et al*. (1999) applied the Huang and Goltz (1998) model to optimize HFTW operating parameters with the goal of minimizing cost and meeting downgradient regulatory standards. To accomplish this, Garrett *et al*. (1999) used genetic algorithms (GAs). Inspired by the processes of natural selection and evolution, GAs maintain and "evolve" in order to minimize an objective function.

2.4.3.2 Modeling Fractured Media

As was shown in Section 2.4.1, the prognosis for cleanup of NAPL contaminants in fractured systems is extremely poor, so containment of the contaminant plume becomes the remedial goal (Mackay and Cherry, 1989). Since pump-and-treat is the remediation technology of choice for plume containment, we want to be able to simulate the effect of pumping systems on groundwater flow in a fractured aquifer system.

Fractured aquifers generally contain cracks (fractures) of various lengths, widths, and apertures. The aquifers are permeable primarily because of the effective porosity provided by these fractures rather than that of the relatively impervious rock matrix, as is shown in Figure 2.10. Additionally, large fractures or fissures can behave like channels. The effective fracture porosity of fractured-rock aquifers is generally between .001-10%, much smaller than the porosities of typical porous media aquifers (20-40%).

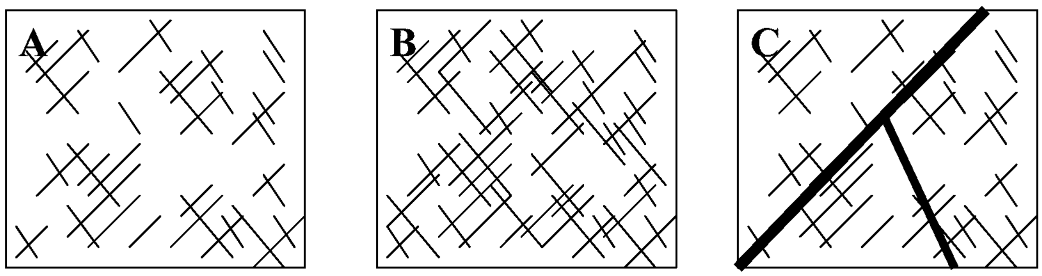


Figure 2.10: Porosity differences in fractured media. (a) Unconnected fractures with minimal flow, (b) connected clusters that would support fractured flow, and (c) large fractures dominating the matrix inducing channel flow (From Domenico and Schwartz, 1998)

When determining the permeability of fractured media in the ROK, the fracture density is important in relation to the fracture connectivity. Percolation theory deals with how fracture connectivity and density promote flow. A fractured media can have a high fracture density, but if the fractures are not connected, flow is prevented. Conversely, a high degree of connectivity between fractures promotes flow. When the fracture density becomes sufficiently high, isolated fractures become rare, and the percolation threshold for flow is reached. It is at this stage that the aquifer behaves as a porous media (Domenico and Schwartz, 1998; NRC, 1990).

Equivalent porous medium (EPM) models (Long *et al*., 1982; Pankow *et al*., 1986; Schmelling and Ross, 1989) assume that at the scale of interest the fractured aquifer behaves identically to an unconsolidated medium. In order to determine the scale of interest, a representative elemental volume (REV) is sought for various parameters (NRC, 1990; Lee *et al*., 2001). If considering the parameter hydraulic conductivity (K) for example, the REV is found when a small change in the averaging volume does not result in a notable change in K. The concept of selecting an REV is illustrated in Figure 2.11. In a small sample where the porous area is small, very small changes in the averaging volume can cause appreciable changes in the K. However, when the sample becomes very large, the K will no longer be sensitive to the averaging volume.

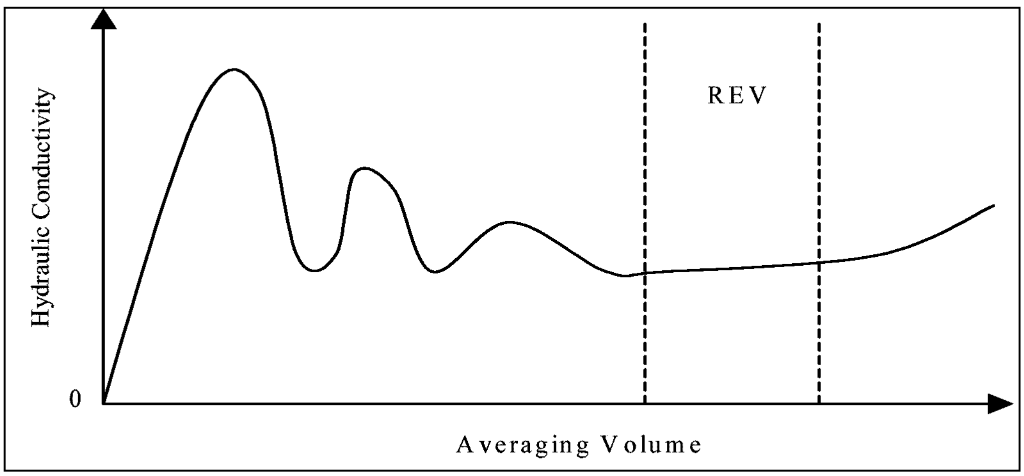


Figure 2.11: Variation in hydraulic conductivity (K) as a function of the averaging volume. The dashed lines point to volume where the assumption of an REV is valid (From NRC, 1990)

EPM has been used with success in the past and for most cases it models the response of fractured systems adequately for design purposes (NRC, 1990; EPA, 2001b). Note though, that the EPM is applicable only when the volume sampled is sufficiently large (Gernand and Heidtman, 1997; Lee and Lee, 1998). For instance, assume we want to use an EPM approach to model drawdown in a pumping test. At small sampled volumes, flow through individual fractures may substantially affect overall flow, resulting in non-symmetric drawdown around the well. However, as the sampling volume is expanded, the drawdown will be symmetric about the well, as it would in a homogenous porous medium (Figure 2.12). If a fractured aquifer can be characterized as an EPM, then pumping test data may be interpreted by methods such as Theis curve matching, which were developed for porous systems.

Lee and Lee (1998) applied the EPM to characterize flow in a fractured media weathered gneiss system in Wonju, ROK. The authors cite three conditions that should be satisfied in order to use an EPM model to analyze an aquifer: (1) small fracture spacing (high fracture density), (2) high fracture connectivity, and (3) random fracture orientation. To test whether these criteria were met, rock cores were obtained from numerous wells. No dominant fractures were found in the cores, and a borehole imaging processing system revealed high fracture densities and equal distribution of fractures. Pumping test data closely matched Theis drawdown curves, and there were no indications of single fracture flow or double porosity behavior (discussed later in this section). Lee and Lee (1998) concluded that the choice of an EPM model was valid for the Wonju system. They also noted problems with applying single fracture or double porosity models to real field situations, due to an inability to reliably measure parameters required for the models, making application of such models impractical.

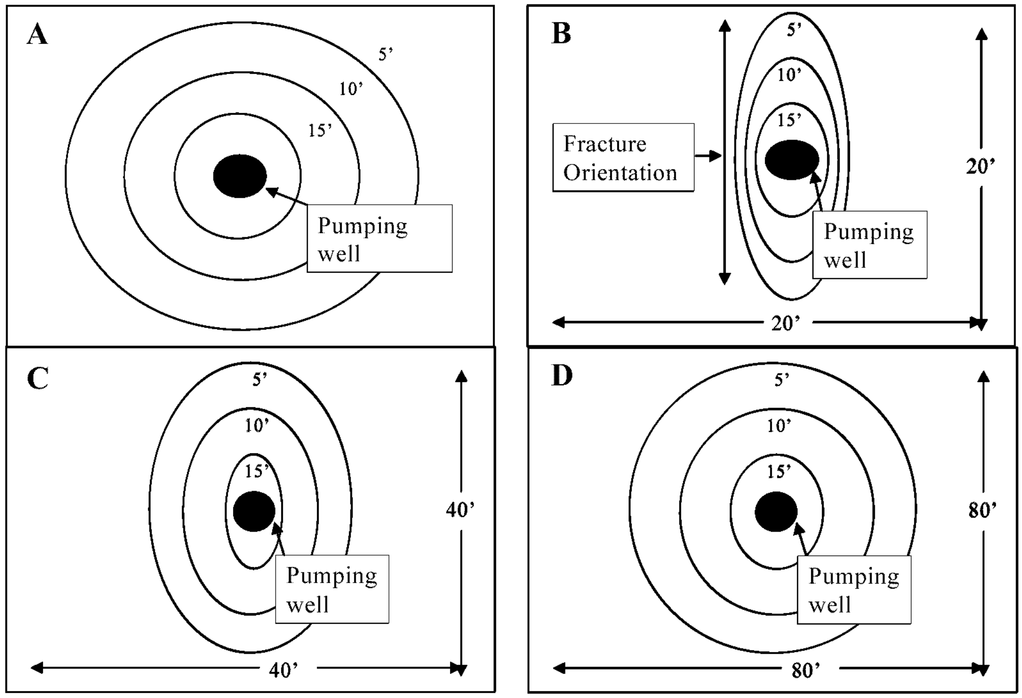


Figure 2.12: Drawdown curves differ for porous media versus fractured media, but begin to converge as measurements are taken farther from the pumping well: (a) drawdown expected from homogenous porous media, (b) drawdown due to fractal flow at small radial distance, (c) and (d) as the radial distance from the pumping well increases, the drawdown begins to resemble that of porous media.

Single fracture models and discrete fracture approaches (Gringarten and Witherspoon, 1972; Gringarten, 1982; Karaski, 1986; NRC, 1990) assume pumping wells are intersected by a single fracture that is significantly more transmissive than the rest of the aquifer (Figure 2.10c). These models characterize the transmissive fractures from drawdown data at a production well. These data typically plot as straight lines on log-log scales at early time, and merge with a Theis curve if the test is sufficiently long (Gernand and Heidtman, 1997). In order to obtain accurate fracture orientations from rock cores, it must be assumed that there are no faults or folds present (Gernand and Heidtman, 1997). Marquis *et al.* (1994) and Gernand and Heidtman (1997) reported that radial flow can be simulated with single fracture models via interconnected fractures from pumping wells, although the distance for it to develop depends on the type of fractured media. At one test site where low-porosity crystalline bedrock was present, Gernand and Heidtman (1997) analyzed rock cores, and identified five fracture sets of various orientations, with no fracture set comprising more than 28 percent of fractures. This led to the assumption that radial flow might develop via the interconnected fractures. After pumping tests were accomplished, it was discovered that radial flow could be observed at observation wells 300 feet from the pumping-well (Figure 2.12d), but that the observation wells within 50 feet (Figure 2.12b) of the pumping-well exhibited characteristics of non-symmetrical drawdown due to flow in a single large fracture (Gernand and Heidtman, 1997).

The channel network model is a variation of the single fracture model that takes into account field observations that fracture surfaces are uneven and mineralized. This heterogeneity causes the flow and contaminants to be distributed non-uniformly across the fracture plane in preferential paths, or channels (Selroos *et al*., 2001). Mixing of flowing water that occurs within the three-dimensional rock matrix (see Figure 2.13) is accounted for by a computer code (CHAN3D), which solves for the flow in each channel in the system using the finite-difference method (Selroos *et al*., 2001).

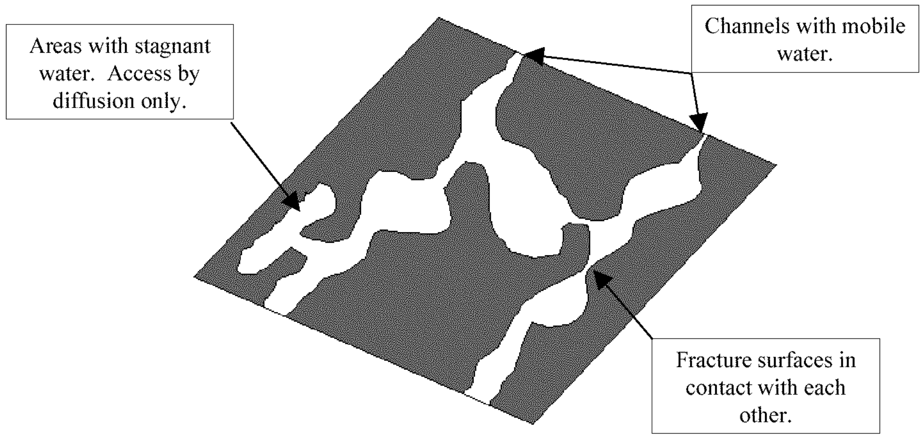


Figure 2.13: **Channel network conceptual model (Selroos *et al.*, 2001)**

The dual-porosity model has also been used to characterize flow in fractured media. The dual-porosity model assumes the simultaneous existence of two distinct porous systems with different values for porosity and permeability. The more permeable system, the fractures, transmits groundwater to the well, while the less permeable media, the rock matrix, has a high storage coefficient and acts as a source (Hamm and Bidaux, 1996; Lee *et al.*, 2001). Unlike the single fracture model, no account is taken of the arrangement of fractures and their relation to one another; instead it is assumed that there is a mixing of fluids in interacting continua (NRC, 1990). When developing this model, flow equations are written for both the rock matrix and the fracture system. The systems are interconnected, so that the loss of fluid in one porous system represents a gain in the other (see Figure 2.14).

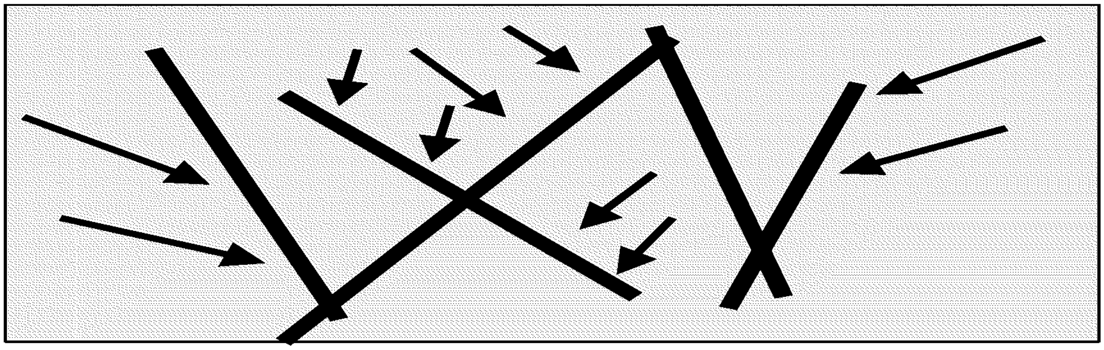


Figure 2.14: As the fractures lose water, capillary forces in the rock matrix are overcome. The water stored in the rock matrix is then released to the fractures.

Another technique for determining aquifer properties is the stochastic approach. An aquifer is considered homogeneous when properties such as conductivity do not vary from point to point. However, homogeneity is an ideal approximation that does not exist in nature (Domenico and Schwartz, 1998). This is due, for example, to stratification within the aquifer (Figure 2.15). When dealing with porous media, in order to determine the aquifer's conductivity on a larger scale, a sampling pattern is established to measure a number of data points for conductivity. These data may be displayed as a histogram (Figure 2.16a). It has been found that when the logarithms of the conductivity data are plotted, the distribution normalizes (Figure 2.16b). Thus, this normal distribution of log conductivity data for an aquifer can be described by specifying a mean and variance. With these statistics specified, a stochastic model of the aquifer can be constructed and used to simulate probabilistic distributions of contaminant.

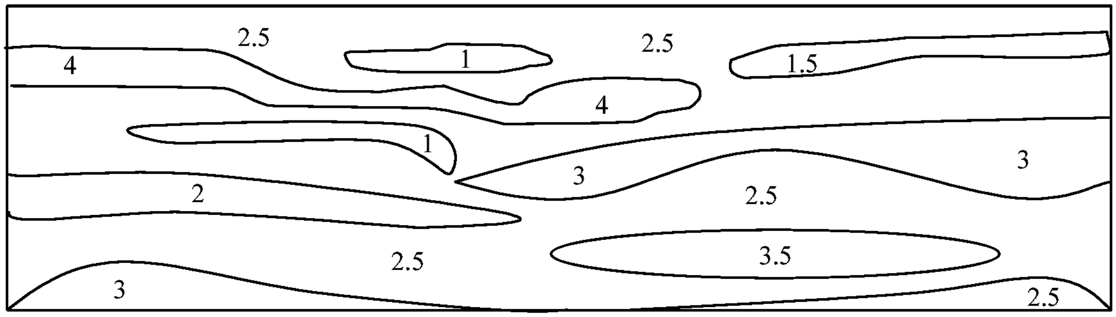


Figure 2.15: Notional hydraulic conductivity (K) distribution (numbers represent the negative log K) for a section of aquifer (From Domenico and Schwartz, 1998)

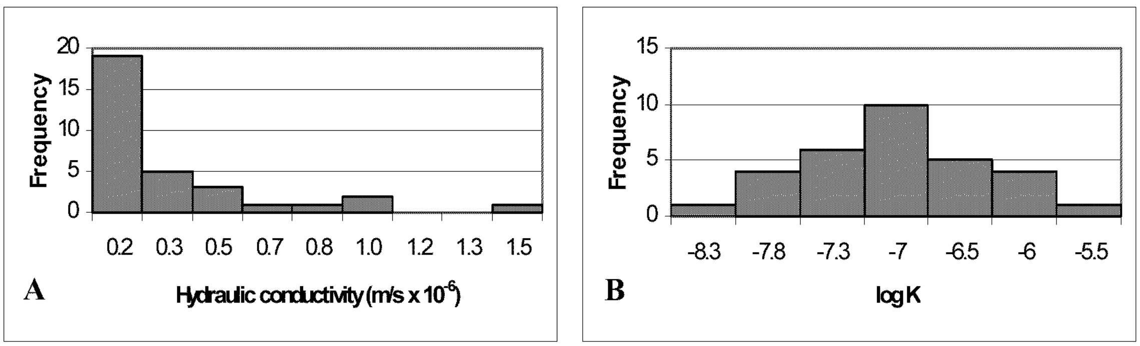


Figure 2.16: (a) Frequency distribution of hydraulic conductivity (K), (b) histogram of the log-transformed hydraulic conductivity data (From Domenico and Schwartz, 1998)

To apply the above-described stochastic approach to a fractured system, it is necessary to assume that the fractured media can be described as an equivalent porous media. Once that assumption is made, the steps involved in determining the descriptive statistics of the fractured medium, and then developing a stochastic model, are identical to those described above for a porous medium. Selroos *et al*. (2001) used this approach to simulate groundwater flow and radionuclide transport in fractured rock. They then compared the stochastic modeling approach with modeling flow in discrete fractures and modeling flow in a channel network, and found that the three approaches gave similar results.

3 Methodology

3.1 Introduction

In Chapter 2, it was shown that fractured media systems are typical in the ROK and that frequently chlorinated solvents contaminate these systems. We also showed that based on cost, safety, and efficiency, HFTWs are a good technology to use to deal with chlorinated solvent contamination, especially when the water table is relatively deep (as is many times the case in the ROK). The question now arises, can these HFTWs be effectively applied under the hydrogeological conditions typically encountered in the ROK. To answer this question, a model of HFTW operation in fractured media is useful.

In this chapter, we formulate such a model. First we present our model assumptions and justify the use of existing HFTW models, which have been used to simulate HFTW operations in porous media, to describe HFTW operations in fractured media. In the second section of the chapter, we develop a contaminated site scenario based on contamination and hydrogeology at Osan AB. In the third section of the chapter, we describe the sensitivity analyses that will be conducted, with results presented in Chapter 4, to answer the research question: how can HFTW technology be applied at USFK installations in Korea?

3.2 Modeling HFTW Operation in Fractured Media

In the first part of this section, we justify use of an equivalent porous media model to simulate HFTW operation at a contaminated fractured media system in the ROK. We then select from between the two HFTW models discussed in Chapter 2, analytical and numerical, for further application.

3.2.1 Assumptions

1) EPM: Our first assumption is that an EPM model is appropriate for use in simulating an HFTW system to remediate contaminated fractured media at Osan AB. The specific fracture density, orientation, and aperture at Osan AB are unknown, and in general, difficult to measure without considerable expenditure of site characterization funds—funds which would typically be unavailable at DoD installations overseas. Since we have seen that these data would be required to apply dual-porosity, single fracture, or stochastic groundwater flow models, we may conclude that application of such models would be impractical at Osan AB in particular, and overseas installations in general. We have also seen that in the absence of detailed fracture characterization, EPM models have been successfully used (Lee and Lee, 1999). Lee and Lee (1999) justified their use of the EPM approach in a weathered gneiss aquifer in Wonju, ROK, when modeling a system with relatively high fracture density. The United States Army Corps of engineers assumed an EPM when conducting a site survey at Wonju and determining hydraulic conductivity (USACE, 2001).

Sources (Lee and Lee, 1999; Osan AB CE, 2001; USACE, 2001) have suggested that weathered gneiss is common throughout the ROK. In particular, the well-bore data from Osan AB (Appendix 1) indicate highly fractured weathered gneiss. Thus, the EPM approach taken by Lee and Lee (1999) at Wonju would also seem to be applicable at Osan AB, and presumably, many other sites in the ROK. The key to using an EPM model is that the scale of the fractures be small in comparison to the scale of the system being simulated (Lee and Lee, 1999; Long *et al*., 1982; Pankow *et al*., 1986; Schmelling and Ross, 1989). Wells at Osan AB have been drilled deep (100-300 meters) into weathered gneiss that has a high fracture density (see section 2.2.3), and the contaminant plume at Osan AB has traveled hundreds of meters (see Figure 3.1 and Table 3.1). Fracture spacing, length, and aperture in weathered gneiss are often measured in millimeters or centimeters (Charbeneau, 2000; Domenico and Schwartz, 1999; EPA, 2001; NRC, 1990). The Army Corps of Engineers assessment at Wonju reported fractures there were typically 10mm wide (USACE, 2001). Therefore, it is assumed that the scale of interest (tens or hundreds of meters) versus the scale of fracturing (centimeters) is large enough to support the use of an EPM.

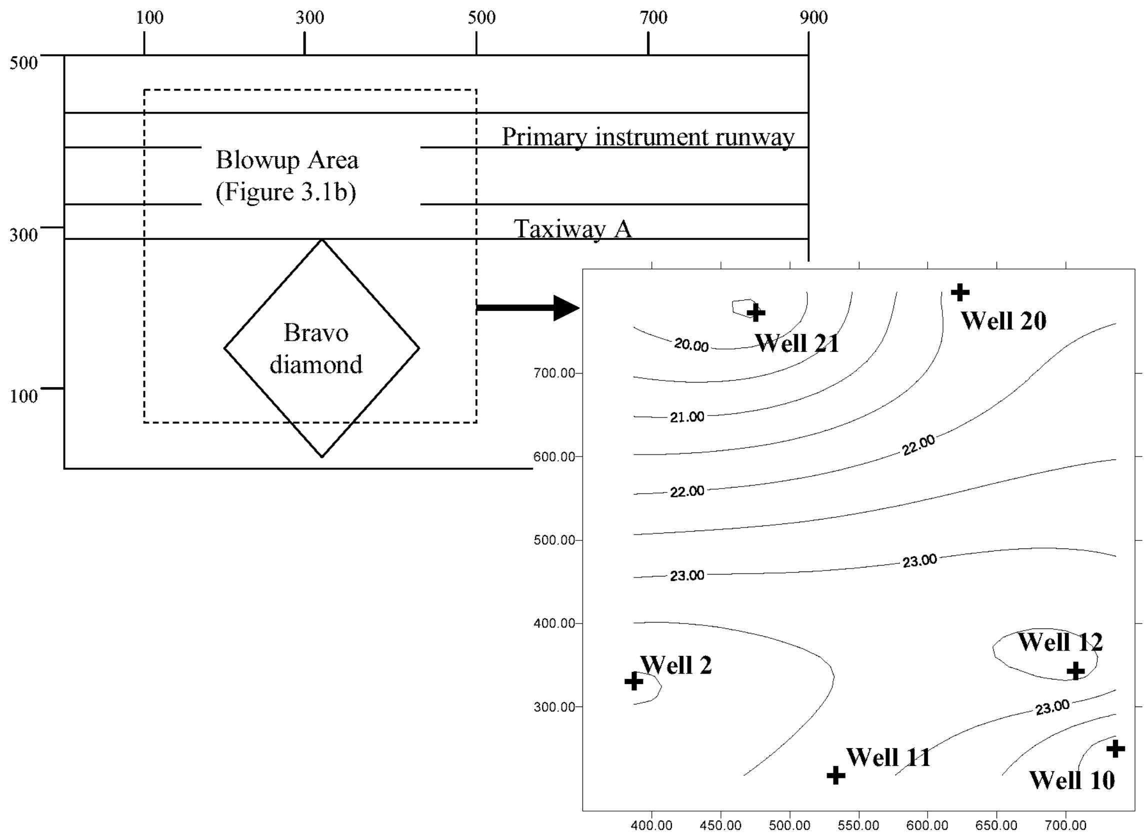


Figure 3.1: Site map of Osan AB (scale and contours in feet) showing area of interest with well location and hydraulic head contours

Table 3.1: Contaminant levels and non-detects (ND) observed in Osan AB wells

	Well #					
	2	10	11	12	20	21
TCE (ug/L)	16.3	34.4	15.1	ND	30	ND
DCE (ug/L)	26	72.2	ND	1.1	64.7	ND
VC (ug/L)	ND	19.6	ND	10.8	17.6	ND

2) Horizontal Layers: A second assumption is that we can model the Osan AB aquifer as a horizontally layered system. Well borings taken from Osan AB show four distinct layers (Table 3.2) of differing soil composition. Comparisons of the borings show the layers are generally found at the same depth for the region being modeled (Appendix 1). Thus, we assume we can model hydraulic conductivity at Osan AB using a minimum of four layers. The layers may have different values of hydraulic conductivity and conductivity in the vertical and horizontal directions may exhibit anisotropy, though we will assume the conductivity and anisotropy within each layer is homogeneous.

3) Steady-State Flow: As the modeling focuses on the long-term containment of a chlorinated solvent plume, a steady-state flow field for the region and HFTW system was assumed since the time-scale of transient fluctuations in flow is small compared to the time-scale of the contaminant transport. This steady-state flow assumption is typically made when modeling contaminant transport by groundwater (Bakker and Strack, 1996; Bumb *et al*. 1997; Charbeneau, 2000).

4) Biodegradation and Sorption: As noted in Chapter 2, the natural attenuation of chlorinated solvents is not necessarily an important process. As there is no clear evidence of significant biodegradation in the Osan AB plume, we will take a conservative approach and assume biodegradation is negligible. McCarty *et al*. (1998) reported in the Edwards test site that sorption was negligible as well, due to low fractions of organic

carbon present in the aquifer. It is likely that the deep fractured gneiss at Osan AB will have similarly low fractions of organic carbon, so we will also assume sorption is negligible.

3.2.2 Model Selection

In order to adequately simulate an aquifer as a system of anisotropic layers, unless we make some very severe simplifications, a numerical model is required. Of the two numerical models discussed in Chapter 2 that have been used to simulate HFTWs, the model by Huang and Goltz (1998) is readily accessible by the author, with technical support easily available. For these reasons, the numerical model by Huang and Goltz (1998) will be applied to the problem.

3.3 Site and Technology Model

In this section we will construct a simple model of the TCE contamination at Osan AB. In the first part of this section, model parameters and assumptions are explained. We then move on to model verification. In the following section, we will set up a sensitivity analysis in which we allow some model parameters to vary, in order that we may simulate how the technology performs under different conditions.

3.3.1 Model Parameters and Detailed Assumptions

The simplified site at Osan AB is modeled with dimensions 200 meters wide, 200 meters long, and 120 meters deep. The MODFLOW finite difference grid is 100 cells wide by 100 cells long by 12 cells deep, so that each cell is 2 meters wide, 2 meters long, with varying depths, as listed in Table 3.2. Huang and Goltz (1998) suggest the finite difference grid design should have a cell ratio (length by width by depth) that does not exceed 1x1x8 in order to get accurate results. As such, the weathered gneiss was separated into layers 10 meters thick, resulting in a 1x1x5 ratio. The parameters and associated assumptions used when creating the model are described below:

1) Hydraulic Conductivity: Values (Table 3.2) for overburden conductivity and porosity were found in the literature (Envirobrowser, 1998; Charbeneau, 2000). The literature reports a wide range of conductivity values for weathered and biotite gneiss, as noted in the table (Envirobrowser, 1998).

Table 3.2: MODFLOW layer design

Layer	Media Type (layer depth)	K (cm/s)	Porosity
1	Brown Clayey Sand (7m)	2.55E-05	0.3
2	Pale brownish gray weathered sericite gneiss (13m)	6.94E-04	0.39
3	Gray to pale brownish gray weathered gneiss (10m)	10^{-3} - 10^{-5}	0.1
4	Gray to pale brownish gray weathered gneiss (10m)	10^{-3} - 10^{-5}	0.1
5	Gray to pale brownish gray weathered gneiss (10m)	10^{-3} - 10^{-5}	0.1
6	Gray to pale brownish gray weathered gneiss (10m)	10^{-3} - 10^{-5}	0.1
7	Dark brownish gray biotite gneiss (10m)	10^{-3} - 10^{-8}	0.1
8	Dark brownish gray biotite gneiss (10m)	10^{-3} - 10^{-8}	0.1
9	Dark brownish gray biotite gneiss (10m)	10^{-3} - 10^{-8}	0.1
10	Dark brownish gray biotite gneiss (10m)	10^{-3} - 10^{-8}	0.1
11	Dark brownish gray biotite gneiss (10m)	10^{-3} - 10^{-8}	0.1
12	Dark brownish gray biotite gneiss (10m)	10^{-3} - 10^{-8}	0.1

Drawdown data were available for two wells at Osan AB in the region of interest. These data were analyzed using the Theis curve fit method to determine conductivities appropriate for modeling the gneiss. A Theis curve fit for Well 2 (Figure 3.1) at Osan AB is shown in Figure 3.2. Well 2 is 380 feet deep (116m), has a well radius of 4 inches, and was pumped at 75 gallons per minute for 27.5 hours. The borehole for well 2 is cased and grouted to 65 feet (20m) below surface. Below the first 65 feet, the gneiss is used as a natural screen for the additional 315 feet (99m). Thus, in essence, the well is screened over layers 3 through 12. As the drawdown data do not support calculating the biotite and weathered gneiss conductivities separately, the conductivity found from the

Theis curve is an average. The curve fit yielded a hydraulic conductivity of 6.32×10^{-4} cm/sec.

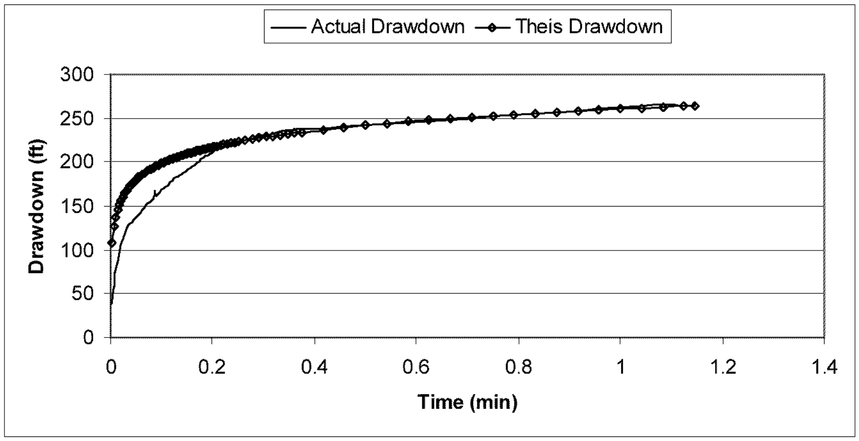


Figure 3.2: Theis curve fit of drawdown data at Osan AB, ROK

Differences between early drawdown readings (less than 400 minutes) are attributed to well bore storage (Driscoll, 1986; Johnson, 1966; Lee and Lee, 1998). The Theis curve fit yields a hydraulic conductivity within the range of values for gneiss in Table 3.2, suggesting hydraulic conductivities for the model are on the order of 10^{-4} cm/sec.

2) Constant head boundaries: As discussed in section 3.2.1, the model assumes steady state flow conditions. As such, constant head boundaries were defined to induce flow across the region. Groundwater elevation readings taken from well development logs (Appendix 1) and hydraulic contour mapping were used to estimate a regional hydraulic gradient of .00125. This hydraulic gradient was imposed by specifying a constant head at the left boundary of 6.5 meters below ground surface, while the constant head at the right boundary (200 meters away) was specified at 6.75 meters below ground surface (Figure 3.3). Groundwater flow induced by the natural gradient was unidirectional, from left to right. No flow conditions were imposed along the boundaries parallel to flow.

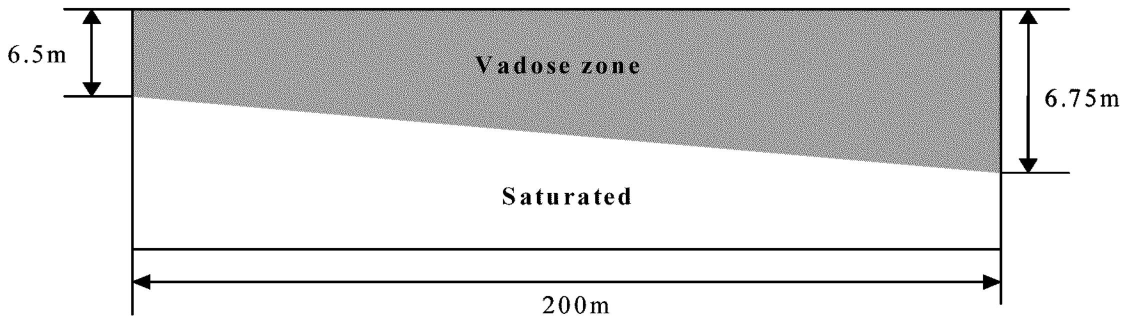


Figure 3.3: Constant head boundaries used to model Osan AB

3) Anisotropy: Anisotropy describes the condition where vertical and horizontal hydraulic conductivity values differ so that groundwater flows with more ease in one direction than the other. Literature reports that anisotropy up to two orders of magnitude is generally found in fractured rock (Domenico and Schwartz, 1999; Charbeneau, 2000). There was no "typical" value for anisotropy in fractured media reported, but a factor of ten is commonly used for porous media (Charbeneau, 2000; McCarty *et al.*, 1998) Therefore, an anisotropy of ten was assumed for the model.

4) Two-well HFTW system: The number of wells used for the application of HFTW technology is determined so that the entire plume can be captured and adequately treated by the system. To simplify the modeling in this study, a plume was generated (as discussed below) so that it could be treated using a two-well HFTW system. Managing larger plumes would merely involve adding more wells.

5) Treatment Efficiency: The model assumes that with each pass of contaminated water through a treatment well, 90% of the contaminant will be destroyed. This is consistent with previous applications of HFTW in the field (McCarty *et al.*, 1998).

6) Pumping Rate: MODLFLOW was used to determine the maximum pumping rates that could be used in the model without dewatering the layers. MODFLOW displays

dewatered cells as “dry” cells when the pumping rate exceeds the maximum pumping rate. The maximum pumping rate for the model was determined to be 1 m^3/day per layer, and is the base pumping rate for the model.

7) Plume generation: The plume was generated assuming the contaminant source had migrated through the overburden and into the weathered gneiss to the full well depth of 120 meters. As the highest TCE concentration measured at the site was 83 μg/L, a value of 100 μg/L was used for the plume concentration at the left boundary of the site.

To determine the plume width that could be handled by two HFTW treatment wells pumping at 1 m^3/day per layer, an analysis was conducted comparing the mass entering the system from the source cells with the mass removal capabilities of the HFTW system. The constants used in calculating the mass entering the system from the source cells are shown in Table 3.3, while Table 3.4 summarizes the rate of contaminant mass entering the system per 10 m deep layer (D = 10m) for various source widths, where source width is calculated as number of source cells in a layer (n) times cell width (W = 2m).

Table 3.3: Aquifer Constants

Hydraulic Conductivity (K)	6.32E-06	m/s
Plume Source Concentration (C)	0.1	mg/L
Hydraulic Gradient (I)	0.00125	

Table 3.4: Rate of contaminant mass entering the system per layer as a function of plume width

		Plume Width (n)							
	Units	**2**	**4**	**6**	**8**	**10**	**20**	**30**	**40**
Cross sectional area of source (A=WDn)	m^2	40	80	120	160	200	400	600	800
Darcy Velocity (q=KI)	m/s	7.90E-09	7.90E-09	7.90E-09	7.90E-09	7.90E-09	7.90E-09	7.90E-09	7.90E-09
Flow (Q=qA)	m^3/day	0.027	0.055	0.082	0.109	0.137	0.273	0.410	0.546
Contaminant transported from source (C*Q)	mg/day	2.73024	5.46048	8.19072	10.921	13.6512	27.3024	40.9536	54.6048

The maximum destruction of contaminant per pass of water through an HFTW treatment well was set at 90%, and the amount that could be destroyed per layer was determined (Table 3.5).

Table 3.5: HFTW mass destruction per layer

HFTW Water flow per layer per day	1.0	m^3/day
Rate of contaminant mass treated per layer (@ max concentration of 0.1 mg/L)	100	mg/day
Rate of contaminant mass destroyed per layer (@ 90% efficiency)	90	mg/day

The greatest contaminant mass the HFTW system can destroy is 90 mg/day per layer. Even when the source width is 60 cells wide (54.6 mg/day), the treatment capacity of a two well system is not exceeded. Trial runs were conducted to determine an appropriate plume source width. These trials indicated that a plume source width of 60 (120 meters) to 6 cells (12 meters) would not be captured by an HFTW system placed 39 meters downgradient of the source (Figure 3.4a). Therefore, a plume source width of 4 cells (8 meters) was chosen to ensure the plume could be captured by a two-well system (Figure 3.4b).

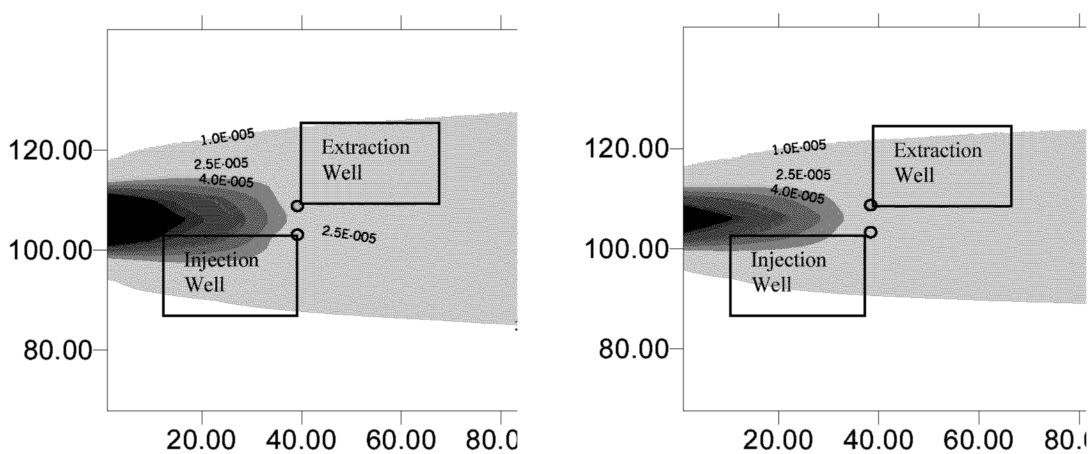


Figure 3.4: Contaminant plume at steady state for layer 5 with (a) plume source width of 12 meters and (b) plume source width of 8 meters

8) Screen interval: The HFTW treatment wells were screened over all the saturated layers (2-12) with the exception of layer 7 (Figure 3.5).

A plan view of the model site used for the modeling effort is shown in Figure 3.5a. Two monitoring wells (MW) spaced evenly about the plume centerline are placed 40 meters downgradient of the HFTWs to monitor contaminant concentrations. The monitoring wells measured concentrations for all of the layers (Figure 3.5b). Contaminant concentration data within the HFTW cells were collected also. The HFTW treatment wells penetrated the full depth of the model, 120 meters (Figure 3.5.).

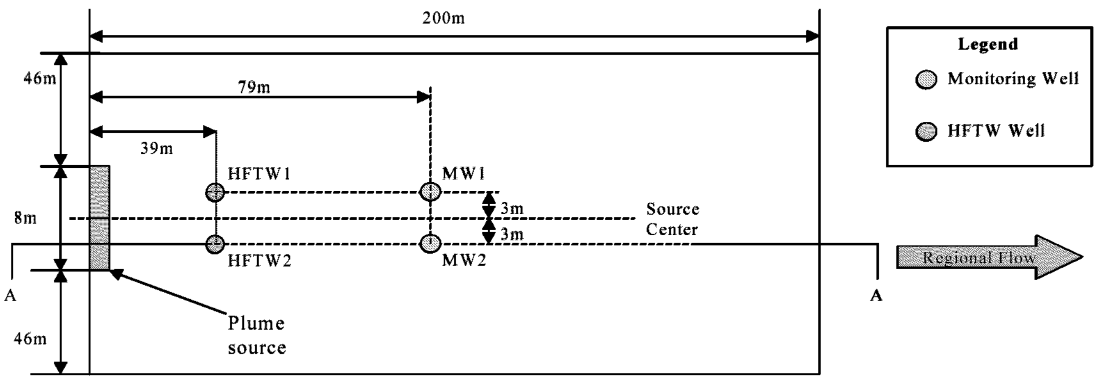


Figure 3.5a: Plan view of model grid

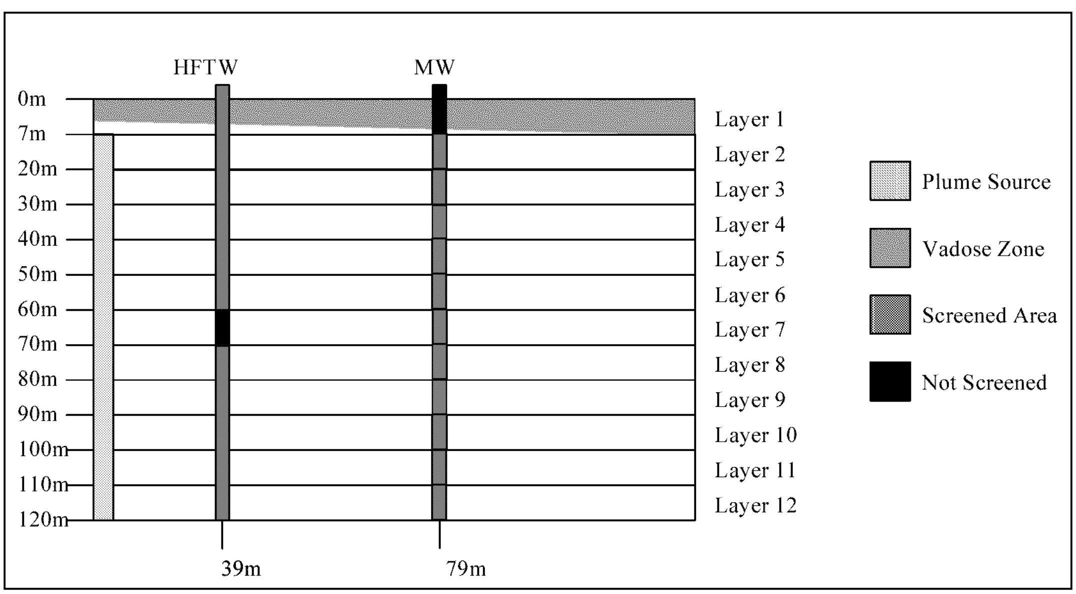


Figure 3.5b: Section A-A

3.3.2 Model Verification

Following the construction of the model, it was necessary to verify that it was operating properly. Two model verification tests were applied: 1) a comparison of modeled and estimated contaminant breakthrough times at monitoring well 1 and 2) a comparison between analytical and numerical model predictions of the HFTW recycle ratio. These tests are described below.

With the two treatment wells not operating, the estimated time for the contaminant to reach monitoring well 1 due to regional flow only was calculated using the average linear velocity of the groundwater (Darcy velocity divided by porosity of .1). The expected time for the contaminant to reach well 1 if the contamination source is activated at day 0 is 11,574 days.

The percent water recycled between the HFTW treatment wells can be determined using the model by comparing contaminant concentration in the treatment well when both wells are operating at 100% contaminant destruction efficiency with the concentration in the treatment well when both wells are operating at 0% treatment efficiency. The analytical model developed by Christ (1997) may then be used to verify the recycle ratio predicted by the numerical model. Christ's model (1997) assumes infinite anisotropy (no vertical flow), and calculates the recycle rate for one layer only. To compare results between the numerical and analytical model, a representative layer was chosen (layer 6), an anisotropy of 10 was used, and the pumping rate was 1 m^3/day per layer.

3.4 Model Analyses

After verifying model performance, we will conduct an analysis of the technology using the model in order to answer the research question of whether the HFTW technology is appropriate for application in the ROK. In the first subsection, we discuss how the treatment system's performance will be measured. In the second subsection, we discuss which engineered and environmental parameters we will vary in order to observe their effect on technology performance.

3.4.1 Measures of Performance

In order to measure how well the HFTW system is performing in containing the contaminant plume, we will compare concentration data from monitoring wells downgradient of the treatment system when the system is operating with downgradient concentrations when the system is not operating.

3.4.2 Analysis

In order to analyze how the HFTW system would be expected to perform in a fractured aquifer, aquifer and system parameters were varied and the impact on performance measured. In the first part of the analysis, we vary aquifer parameters, particularly hydraulic conductivity and anisotropy of the aquifer layers. In the second part of the analysis, we vary technology parameters (pump rate and treatment well screen interval). As noted above, technology performance was assessed by comparing downgradient concentrations simulated with the HFTW system running and with the system turned off.

3.4.2.1 Varying aquifer parameters

The performance of the technology for various environmental conditions was assessed by changing the hydraulic conductivities and anisotropy of the layers. Table 3.6 lists the base case aquifer parameter values. In addition to the base case, six scenarios where aquifer parameters were varied were evaluated. These scenarios are described below.

Table 3.6: Base model parameters

Layer	K (cm/sec)	Anisotropy	Layer	K (cm/sec)	Anisotropy
1	2.55E-05	10	7	6.32E-04	10
2	6.94E-04	10	8	6.32E-04	10
3	6.32E-04	10	9	6.32E-04	10
4	6.32E-04	10	10	6.32E-04	10
5	6.32E-04	10	11	6.32E-04	10
6	6.32E-04	10	12	6.32E-04	10

3.4.2.1.1 Hydraulic Conductivity

As previously noted, the hydraulic conductivity for weathered gneiss varies from about 10^{-3} cm/sec through 10^{-5} cm/sec, and biotite gneiss conductivity can vary over ranges of 10^{-3} cm/sec through 10^{-8} cm/sec. As the hydraulic conductivity being used in the model of the Osan aquifer is only an extremely rough approximation based on geologic characterization, it is of interest to see how the technology would be expected to perform for varying conductivity. To examine this effect, the hydraulic conductivity of layer 7 (not screened by the HFTW system) was changed. In scenario 1, layer 7 was assigned a hydraulic conductivity of 6.32×10^{-2} cm/sec, and in scenario 2, layer 7 was assigned a hydraulic conductivity of 6.32×10^{-6} cm/sec (Table 3.7).

Also of interest was the change in performance of the HFTW technology due to lower or higher gneiss conductivities in layers 3 through 12. To determine this, two additional scenarios were simulated. In scenario 3, layers 3 through 12 were assigned hydraulic conductivities of 6.32×10^{-2} cm/sec, and in scenario 4 layers 3 through 12 were assigned hydraulic conductivities of 6.32×10^{-6} cm/sec (Table 3.7).

3.4.2.1.2 Anisotropy

As mentioned earlier, the literature reports a range of anisotropy for fractured media (Domenico and Schwartz, 1999; Charbeneau, 2000). Therefore, we investigated two scenarios where hydraulic conductivity anisotropy was allowed to vary over two orders of magnitude. In scenario 5, we explore the effects on technology performance when each layer is assumed to be isotropic (anisotropy=1) and in scenario 6 we assign an anisotropy of 100 to each layer (Table 3.7).

3.4.2.2 Varying Technology Parameters

To understand how technology parameters could affect containment of the plume, the pumping rate for each layer was allowed to vary for scenarios 7 and 8. Pumping rate is important, as it establishes the capture zone width and recycle ratio of the HFTWs for given aquifer parameters. In scenario 7, the pumping rate was changed to .1 m^3/day per layer, and the pumping rate was set at .5 m^3/day per layer in scenario 8, with results compared to the base case scenario, where the pumping rate is 1 m^3/day per layer (Table 3.7).

Table 3.7: Summary of hydraulic conductivities (cm/sec), pump rate (m^3/day), and anisotropy values used in model scenarios (changes from the base model are shaded)

		Layer #											
	Property	**1**	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**
Base	Conductivity	2.55×10^{-5}	6.94×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}
	Pump rate	0	1	1	1	1	1	0	1	1	1	1	1
	Anisotropy	10	10	10	10	10	10	10	10	10	10	10	10
Scenario 1	Conductivity	2.55×10^{-5}	6.94×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-2}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}
	Pump rate	0	1	1	1	1	1	0	1	1	1	1	1
	Anisotropy	10	10	10	10	10	10	10	10	10	10	10	10
Scenario 2	Conductivity	2.55×10^{-5}	6.94×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-6}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}
	Pump rate	0	1	1	1	1	1	0	1	1	1	1	1
	Anisotropy	10	10	10	10	10	10	10	10	10	10	10	10
Scenario 3	Conductivity	2.55×10^{-5}	6.94×10^{-4}	6.32×10^{-6}	6.32×10^{-6}	6.32×10^{-6}	6.32×10^{-6}	6.32×10^{-6}	6.32×10^{-6}	6.32×10^{-6}	6.32×10^{-6}	6.32×10^{-6}	6.32×10^{-6}
	Pump rate	0	1	1	1	1	1	0	1	1	1	1	1
	Anisotropy	10	10	10	10	10	10	10	10	10	10	10	10
Scenario 4	Conductivity	2.55×10^{-5}	6.94×10^{-4}	6.32×10^{-2}	6.32×10^{-2}	6.32×10^{-2}	6.32×10^{-2}	6.32×10^{-2}	6.32×10^{-2}	6.32×10^{-2}	6.32×10^{-2}	6.32×10^{-2}	6.32×10^{-2}
	Pump rate	0	1	1	1	1	1	0	1	1	1	1	1
	Anisotropy	10	10	10	10	10	10	10	10	10	10	10	10
Scenario 5	Conductivity	2.55×10^{-5}	6.94×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}
	Pump rate	0	1	1	1	1	1	0	1	1	1	1	1
	Anisotropy	1	1	1	1	1	1	1	1	1	1	1	1
Scenario 6	Conductivity	2.55×10^{-5}	6.94×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}
	Pump rate	0	1	1	1	1	1	0	1	1	1	1	1
	Anisotropy	100	100	100	100	100	100	100	100	100	100	100	100
Scenario 7	Conductivity	2.55×10^{-5}	6.94×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}
	Pump rate	0	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1	0.1
	Anisotropy	10	10	10	10	10	10	10	10	10	10	10	10
Scenario 8	Conductivity	2.55×10^{-5}	6.94×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}	6.32×10^{-4}
	Pump rate	0	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5	0.5
	Anisotropy	10	10	10	10	10	10	10	10	10	10	10	10

4 Results and Analysis

4.1 Overview

In this chapter, we analyze the data from the models described in Chapter 3 to answer the research question: are HFTWs an appropriate technology to remediate contamination under the hydrolgeologic conditions found in Korea. In the first section of this chapter, we verify that the model is working correctly. In the second section, we observe how the model predicts the technology will perform under varying hydrogeological conditions. Finally, in the third section, we examine how the model predicts the technology will perform as we vary engineering design parameters.

4.2 Model Verification

In Chapter 3, we discussed our approach for model verification by comparing: 1) estimated and modeled breakthrough times at a monitoring well and 2) recycle ratios predicted from numerical and analytical models. In Chapter 3, we calculated the estimated time for the plume to reach monitoring well 1, 79 meters from the plume source, with the HFTWs not pumping, based upon the regional hydraulic gradient and applying Darcy's Law. The estimate was 11,574 days. The modeled time versus concentration at this monitoring well for layer six is shown in Figure 4.1.

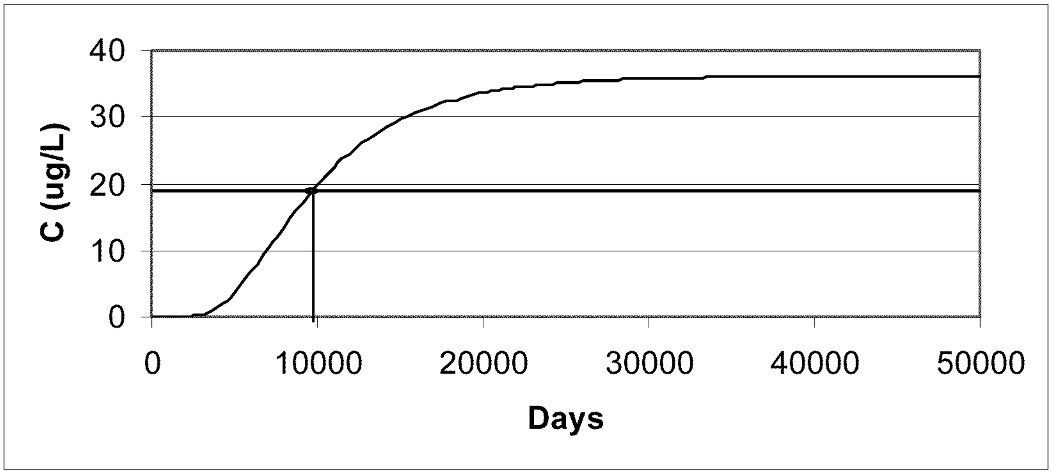


Figure 4.1: Breakthrough curve at layer 6 of monitoring well 1 for contaminant transport by regional gradient only (HFTWs not pumping). Transport time to the well is estimated as the time to attain 50% of the steady-state concentration at the well.

Looking at Figure 4.1, we see the numerical model estimates transport time to well 1 is a little less than 10,000 days. The difference between the numerical model estimate and the estimate from applying Darcy's Law is due to the fact that the numerical model incorporates dispersion. Estimating transport time as the time when the concentration at a monitoring well is 50% of the steady-state concentration will be an underestimate for advective/dispersive transport (Domenico and Schwartz, 1998, pg. 373).

The second part of the model validation involved an examination of the recycle ratio between the wells, when the system was operating with base case parameters (Table 3.7). To calculate this, contaminant concentrations in the treatment wells when both wells were operating at 100% contaminant destruction efficiency were compared with the contaminant concentrations in the wells when the system was operating at 0% efficiency. When both wells were operating at 100% efficiency (Table 4.1), there was a 73% reduction in contaminant concentrations entering HFTW 1 (extraction well) and a 100%

change in contaminant concentrations (clean water) being injected by HFTW 2 (injection well). These data lead to the conclusion that of the water flowing into HFTW 1, 73% comes from HFTW 2, and therefore, the recycle ratio is 73%.

Table 4.1: Steady state concentrations and percent change of contaminant in level 6 of HFTWs 1 (extraction well) and 2 (injection well)

	HFTW 1	HFTW 2
Concentration when HFTW removal efficiency= 0% (ug/L)	36	36
Concentration when HFTW removal efficiency= 100% (ug/L)	10	0
Percent change of contaminant	73%	100%

In order to use the analytical model developed by Christ (1999), the aquifer properties of layer six were used. The depth of the aquifer layer (b) was set at 10 m, the pumping rate (Q) at 1 m^3/day, and the regional Darcy velocity at 6.82×10^{-4} m/day. Christ's model predicted a recycle ratio of 77 %. The slight difference between the analytical and numerical results is likely due to the fact that the analytical model assumes no vertical flow (infinitely anisotropic medium) while the numerical model assumed an anisotropy of 10. Thus, it would be expected that the recycle ratio predicted by the analytical model would be slightly greater than the ratio predicted using the numerical model.

4.3 Modeling sensitivity of treatment system performance to hydrogeologic parameters

4.3.1 Hydraulic conductivity

Two scenarios in which the hydraulic conductivity of layer seven was varied were tested. In scenario 1, the conductivity was raised two orders of magnitude to 6.32×10^{-2} cm/sec. In scenario 2, the conductivity was lowered two orders of magnitude to 6.32×10^{-6} cm/sec. Layer seven was not screened, as it was of interest to examine the impact on technology operation of having high or low conductivity layers not directly captured by the treatment wells. The scenario conditions are summarized in Table 4.2 and breakthrough curves at the monitoring wells (see Figure 3.5 for location) are shown in figures 4.2 through 4.4.

Table 4.2: Series legend for Figures 4.2 through 4.4

Base Case	A	Base case HFTW system off	B
Scenario 1 (High layer 7 conductivity)	C	Scenario 1 HFTW system off	D
Scenario 2 (Low layer 7 conductivity)	E	Scenario 2 HFTW system off	F

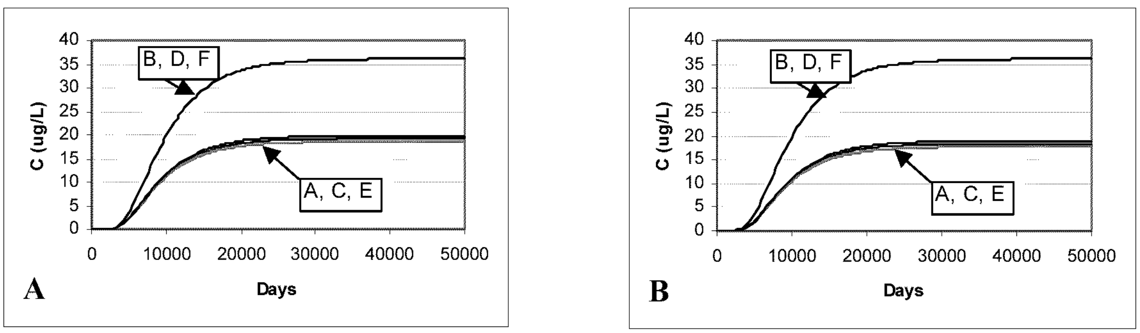


Figure 4.2: Concentration breakthroughs in the 6th layer for varying 7th layer conductivity (a) monitoring well 1 and (b) monitoring well 2

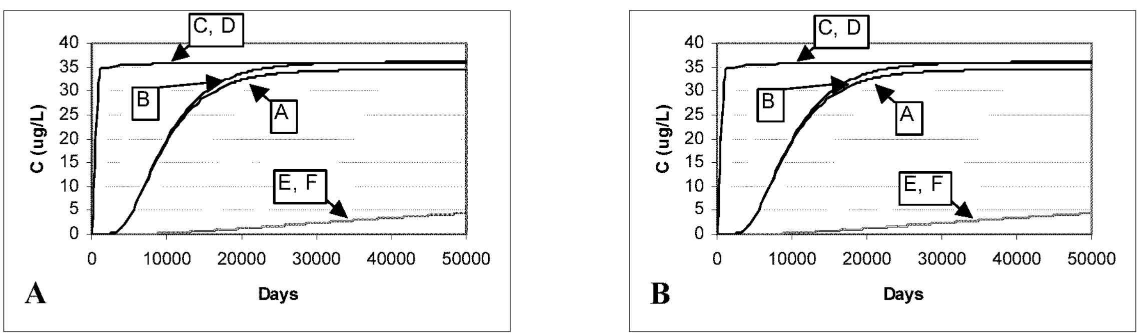


Figure 4.3: Concentration breakthroughs in the 7th layer for varying 7th layer conductivity (a) monitoring well 1 and (b) monitoring well 2

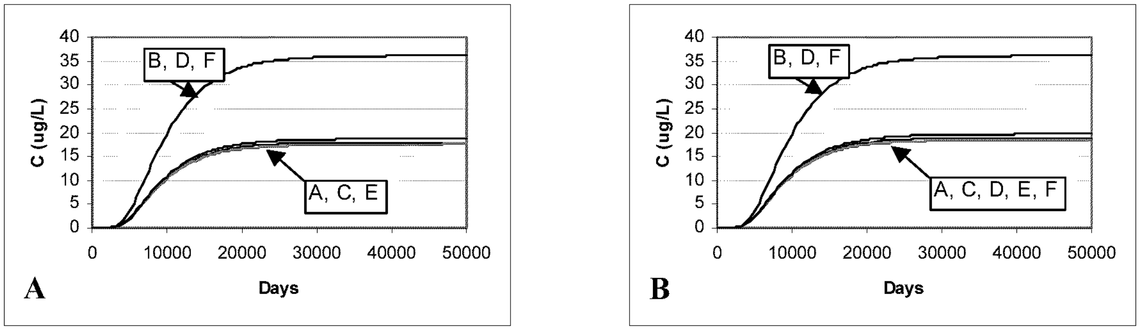


Figure 4.4: Concentration breakthroughs in the 8th layer for varying 7th layer conductivity (a) monitoring well 1 and (b) monitoring well 2

Changes in the hydraulic conductivity of layer 7 had little effect on the treatment achieved in layers 6 and 8. A comparison of the concentration of contaminant at the monitoring wells in layers 6 and 8 shows that they are approximately the same, regardless of the conductivity of layer 7. This is not unexpected, as the aquifer properties and pumping rates in layers 6 and 8 are the same for scenarios 1 and 2. Slight changes in the concentration breakthrough curves are observed in the two layers, presumably due to the effect of vertical water movement into and out of layer 7. However, due to the assumption of a horizontal to vertical conductivity anisotropy ratio of 10, this interaction between layers is relatively small.

The effect of not screening a layer (layer 7) may be seen in Figure 4.3. In scenarios 1 and 2, and the base case, there is little observable change (less than 2%) in the treated (series A, C, and E) and untreated (series B, D, and F) contaminant concentrations in layer 7 once the system reaches steady state. This lack of treatment in a layer that is not screened by the treatment wells could have important implications. As Figure 4.3 suggests, contaminant concentrations in the unscreened layer are unaffected by the treatment system, and, in the case of an unscreened high conductivity layer, contaminants will quickly move downgradient past the system. In scenario 1, for example, a downgradient water supply well intercepting all layers would pump 100 times more water from layer 7 than from any other layer, due to the higher conductivity of layer 7. This increased flow from the untreated layer will result in a higher average contaminant concentration in the pumped water. Scenario 2 does not reach a steady state in the modeled 50,000 days (it would take approximately 140 years to do so), but as the contaminant transport within layer 7 is so slow, the effect of contaminant transport in the untreated layer on a downgradient receptor well would be minimal.

Observable in Figures 4.2 and 4.4 is a slight difference in the contaminant concentrations seen in monitoring wells 1 and 2. This is due to the monitoring wells' locations relative to the HFTWs. In layer 6, monitoring well 1 is directly downgradient of HFTW 1 (extraction), and monitoring well 2 is downgradient of HFTW 2 (injection). Complete mixing has not occurred by the time the plume reaches the monitoring wells, resulting in a contaminant concentration at monitoring well 1 that is slightly higher than the concentration at well 2. In layer 8, which due to symmetry is the mirror image of layer 6, the contaminant concentrations observed at the monitoring wells exhibit the same pattern as in layer 6, but in reverse.

Also of interest is to observe the impact on HFTW performance if the remediation is applied in a very low or high conductivity aquifer. In scenario 3, the conductivity of all the gneiss layers (3-12) was reduced two orders of magnitude to 6.32×10^{-6} cm/sec; and in scenario 4, layers 3 through 12 were assigned hydraulic conductivities of 6.32×10^{-2} cm/sec. Breakthrough curves for the monitoring wells are shown in Figures 4.5 through 4.7 (legend in Table 4.3).

Table 4.3: Series legend for Figures 4.5 through 4.7

Base Case	A		Base case HFTW system off	B
Scenario 3 (Low conductivity for gneiss)	C		Scenario 3 HFTW system off	D
Scenario 4 (High conductivity for gneiss)	E		Scenario 4 HFTW system off	F

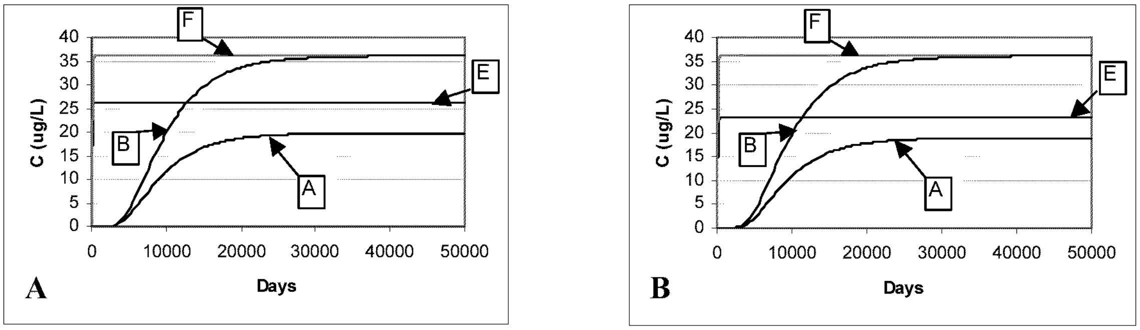


Figure 4.5: **Concentration breakthroughs in the 6th layer for varying gneiss conductivity (a) monitoring well 1 and (b) monitoring well 2**

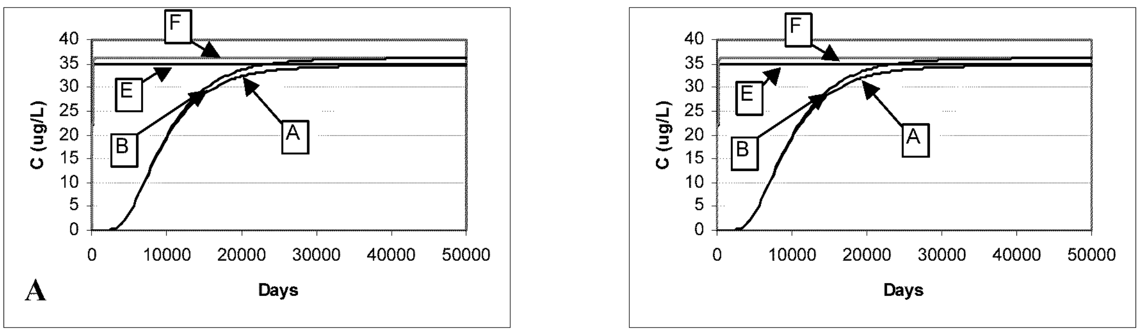


Figure 4.6: Concentration breakthroughs in the 7th layer for varying gneiss conductivity (a) monitoring well 1 and (b) monitoring well 2

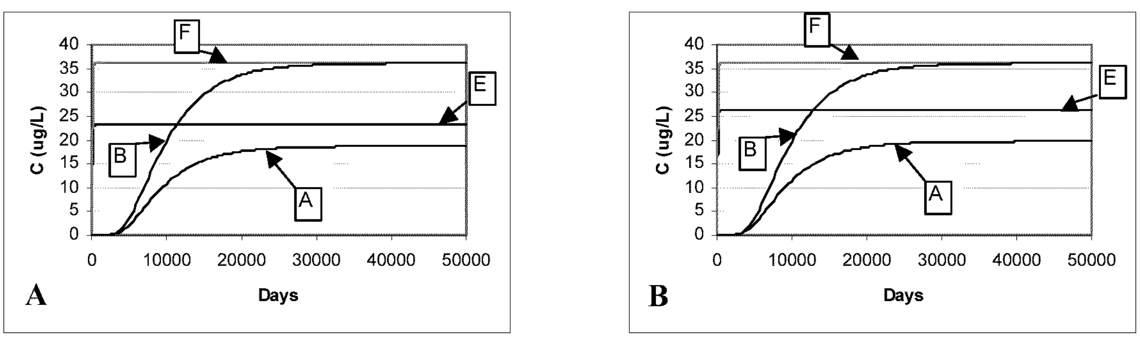


Figure 4.7: Concentration breakthroughs in the 8th layer for varying gneiss conductivity (a) monitoring well 1 and (b) monitoring well 2

Scenario 3 is not shown on the graphs because the low conductivity specified in the scenario could not sustain the HFTW pumping rate of 1 m^3/d per layer.

Excessive water table drawdown resulted in dry cells and meaningless model output. As for scenario 4, the treatment efficiency of the HFTWs for the high conductivity gneiss, based on contaminant concentrations in layers 6 and 8, is less than the treatment efficiency in the base case. This is explained by the fact that the high conductivity of the layers reduces both the recycle ratio and capture zone width, which in turn lowers the system efficiency in removing contaminant. This analysis points out the significant effect hydraulic conductivity has on technology performance, and the importance of basing the technology design on a good estimate of hydraulic conductivity. If the actual conductivity is higher than the design conductivity, system treatment efficiency will be reduced.

4.3.2 Anisotropy

Two scenarios where the aquifer anisotropy was varied were considered. In scenario 5, the model layers were assumed to be isotropic (anisotropy= 1). In scenario 6, the anisotropy in the layers was increased to 100. Breakthrough curves for the monitoring wells are shown in figures 4.8 through 4.10 (legend in Table 4.4).

Table 4.4: Series legend for Figures 4.8 through 4.10

Base Case	A	Base case HFTW system off	B
Scenario 5 (Anisotropy = 1)	C	Scenario 5 HFTW system off	D
Scenario 6 (Anisotropy = 100)	E	Scenario 6 HFTW system off	F

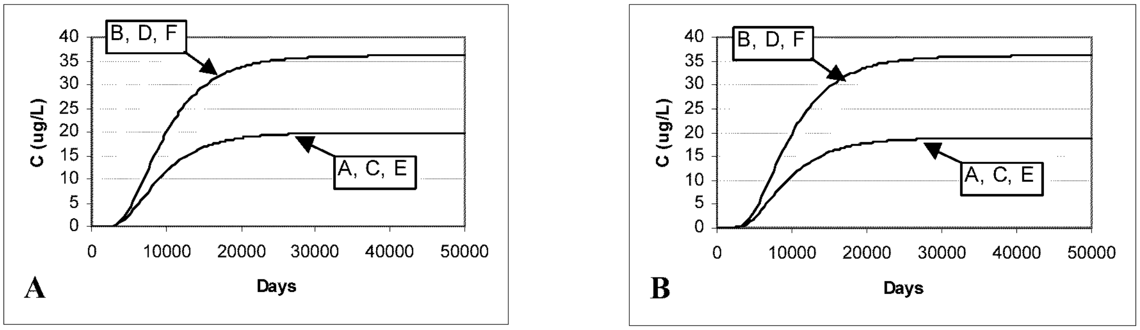


Figure 4.8: Concentration breakthroughs in the 6th layer for changing system anisotropy (a) monitoring well 1 and (b) monitoring well 2

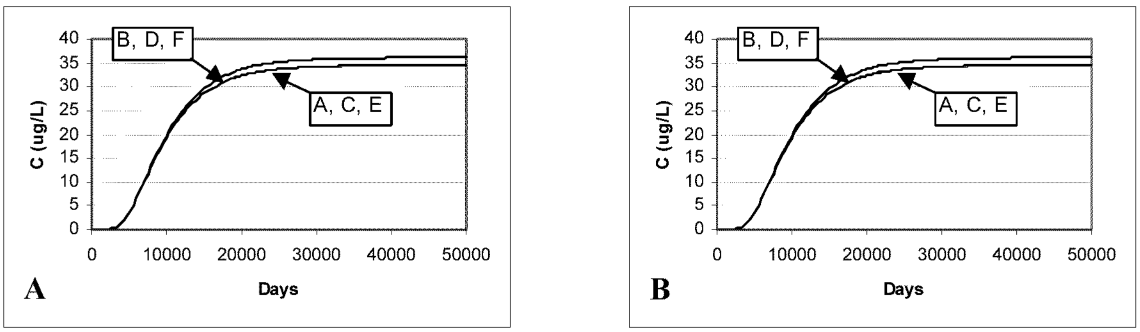


Figure 4.9: Concentration breakthroughs in the 7th layer for changing system anisotropy (a) monitoring well 1 and (b) monitoring well 2

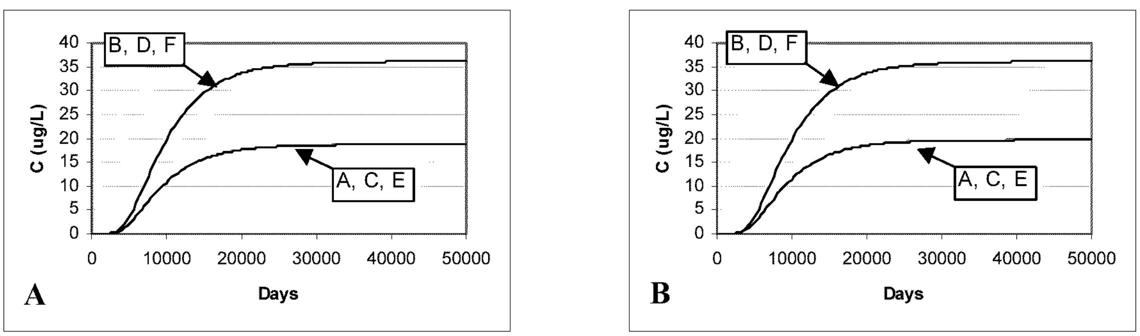


Figure 4.10: Concentration breakthroughs in the 8th layer for changing system anisotropy (a) monitoring well 1 and (b) monitoring well 2

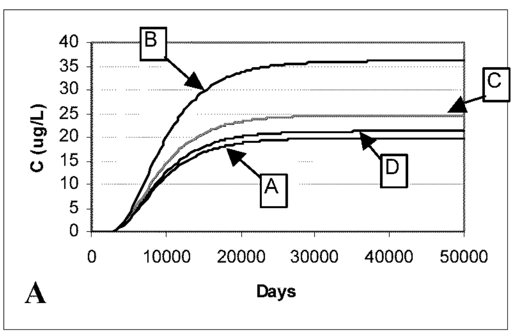
Changes in the layer anisotropy had little effect on the treatment achieved. Whether hydraulic conductivity was isotropic or highly anisotropic, there was no impact on treatment efficiency. This may be due to our assumption that efficiency of the treatment wells is constant. Under some conditions, conductivity isotropy can lead to short-circuiting of water between the screens of a single treatment well, resulting in a loss of treatment efficiency (due, perhaps, to shortening the residence time of contaminated water within *in situ* bioactive zones). As our model assumed constant treatment efficiency, this potential impact of isotropy on treatment efficiency was not observed.

4.4 Modeling sensitivity of treatment system performance to design parameters

The last scenarios executed involved changing the pumping rate of the HFTWs to observe the effects this would have on system performance. In scenario 7, the pumping rate was set at 0.1 m^3/day per layer; while in scenario 8, the pumping rate per layer was set at 0.5 m^3/day. Breakthrough curves for the monitoring wells are shown in figures 4.11 through 4.13 (legend in Table 4.5).

Table 4.5: Series legend for Figures 4.4 through 4.6

Base Case (Pump Rate = 1 m^3/day)	A		Base case HFTW system off	B
Scenario 7 (Pump Rate = .1 m^3/day)	C		Scenario 8 (Pump Rate = .5 m^3/day)	D



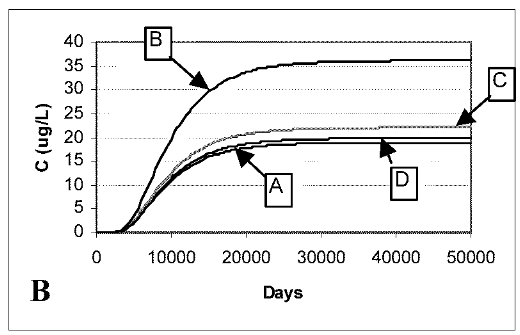


Figure 4.11: Concentration breakthroughs in the 6th layer for changing pump rates (a) monitoring well 1 and (b) monitoring well 2

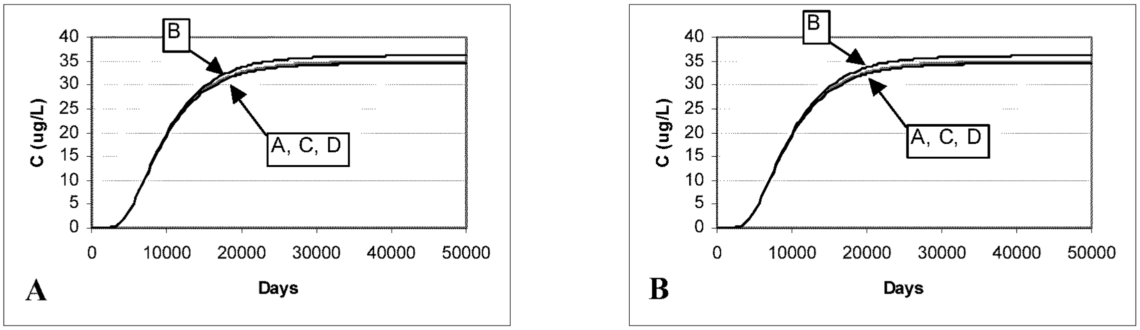


Figure 4.12: Concentration breakthroughs in the 7th layer for changing pump rates (a) monitoring well 1 and (b) monitoring well 2

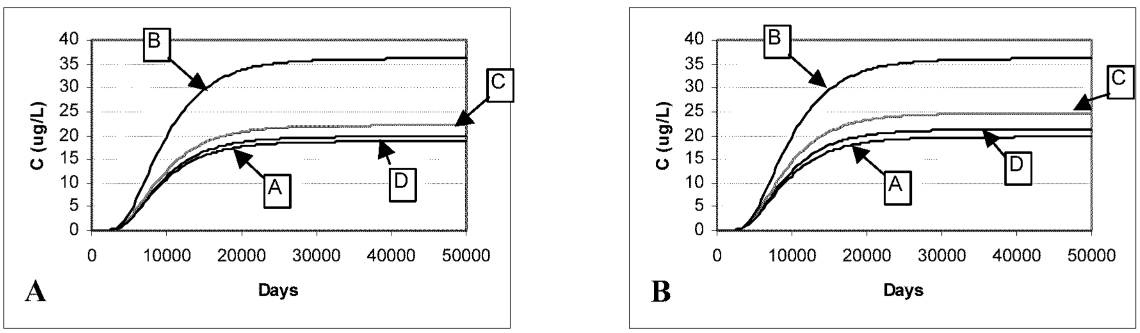


Figure 4.13: Concentration breakthroughs in the 8th layer for changing pump rates (a) monitoring well 1 and (b) monitoring well 2

Figures 4.11 through 4.13 show that technology performance is a strong function of the pumping rate. Downgradient concentrations in layers 6 and 8 are significantly impacted by changes in the treatment well pumping rates. As the pumping rate increases, the CZW and recycle ratio increases, and the contaminant concentration is reduced. Alternatively, as the pumping rate is decreased, the lower CZW and recycle ratio lead to higher downgradient concentrations. Previously discussed (Section 3.3.1) was how the sustainable pumping rate in the system is limited by the hydraulic properties of the aquifers. This suggests that the CZW and recycle ratio between the wells for a given well spacing are limited as well. To increase the recycle ratio, the well spacing would need to be decreased. This reduces the CZW of the system but increases the recycle ratio. To avoid contaminant bypassing the system as a result of the reduced system CZW, additional wells could be added.

The above analyses show the importance of screening over all layers capable of contaminant transport, as well as appropriately sizing and configuring the HFTWs to meet design goals—that is, capturing the contaminant plume and achieving desired downgradient contaminant levels.

4.5 Results Summary and Discussion

In the base case and all of the scenarios modeled, the contaminant concentrations observed 79 meters downgradient of the HFTW system failed to meet MCL limits for trichloroethylene (5 μg/L) or vinyl chloride (2 μg/L). Steady (after 50,000 days) concentrations for the treated plume at the monitoring wells showed treatment approached only 50% removal in the screened layers and 2% removal in the layer not screened (layer 7). The steady state concentrations and percent decrease in contaminant concentration due to treatment observed at monitoring well 1, layers 6 and 7 (monitoring well 2 data were not significantly different) are summarized in Table 4.6. Data from layer 8 are omitted, as the observed concentrations in layers 6 and 8 were approximately the same.

The concentrations above MCLs downgradient of the HFTWs in screened layers can partially be attributed to contaminant bypassing the capture zone of the HFTWs (Figure 4.14). To manage the problem, more HFTWs can be added to increase the system CZW. If necessary to further reduce downgradient concentrations, well spacing can be decreased, which would result in an increased recycle ratio and higher overall treatment efficiency. For the purpose of simplicity in this research, which was focused on analyzing the impacts of aquifer and design parameters on technology performance, the system was limited to consisting of only two treatment wells. , However, for an actual design, where the goal would be to capture a plume and attain contaminant MCLs downgradient, it is likely that the well spacing would need to be reduced and the number of wells increased. Past research (Garrett, 1999) dealt with the optimization of HFTW design parameters to minimize cost and meet downgradient regulatory standards. Decreasing the well spacing and increasing the number of wells in the system increases the overall system cost, but would also reduce the contaminant downgradient of the system (at least in the screened layers).

However, the problem of contaminant transport in unscreened layers on system performance can be observed in layer 7. The contaminant in this layer bypasses the system, with minimal decrease in concentration. To effectively contain a contaminant plume in a fractured system, all layers with conductivities high enough to promote flow would have to be screened. To accomplish this, multiple rows of HFTWs could be installed, and the screen depths of the HFTW rows staggered to ensure the complete depth of the aquifer is intercepted by the system. The additional rows of wells would increase the system cost, but reduce downgradient contaminant concentrations.

Table 4.6: Steady state (50,000 days) concentrations (µg/L) and percent decrease in concentration due to treatment observed in monitoring well 1 for all scenarios

		Treated	Untreated	% Decrease in concentration
Base Case	**Layer 6**	20	35	44%
	Layer 7	35	35	2%
Scenario 1 (high layer 7 conductivity)	**Layer 6**	20	35	44%
	Layer 7	35	35	2%
Scenario 2 (low layer 7 conductivity)	**Layer 6**	20	35	44%
	Layer 7	4	4	1%
Scenario 3 (high conductivity for gneiss)	**Layer 6**	26	36	27%
	Layer 7	35	36	3%
Scenario 4 (low conductivity for gneiss)	**Layer 6**	Invalid	Invalid	Invalid
	Layer 7	Invalid	Invalid	Invalid
Scenario 5 (anisotropy=1)	**Layer 6**	20	35	44%
	Layer 7	35	35	2%
Scenario 6 (anisotropy=100)	**Layer 6**	20	35	44%
	Layer 7	35	35	2%
Scenario 7 (pump rate= .1 m^3/day)	**Layer 6**	25	35	30%
	Layer 7	35	35	1%
Scenario 8 (pump rate= .5 m^3/day)	**Layer 6**	21	35	40%
	Layer 7	35	35	2%

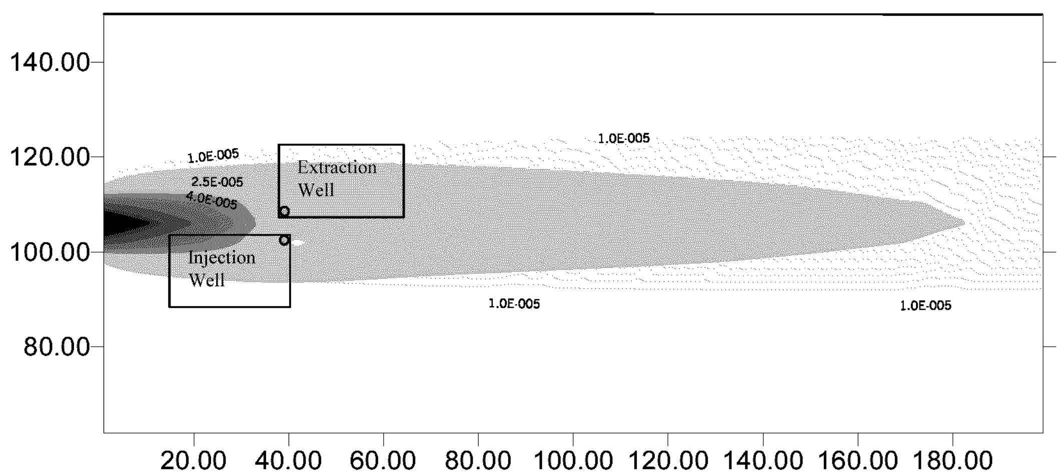


Figure 4.14: Concentration contours (mg/m^3) in layer 6 at 50,000 days. Note that the contaminant plume bypasses the capture zone of the HFTW system, and is transported downgradient

5 Conclusions

5.1 Summary

In this thesis, we analyzed the practicability of using an HFTW treatment system to remediate chlorinated solvent-contaminated groundwater under the hydrogeologic conditions found in the ROK. To perform the analysis, a hypothetical site was constructed based upon conditions at a contaminated site at Osan AB in the ROK, where a weathered gneiss fractured aquifer was found to underlie the base. A model that had been developed for simulation of HFTWs in a porous medium was applied to the fractured medium, after it was shown that it was appropriate to model the fractured system using an equivalent porous medium. Aquifer and technology parameters were then changed to observe the effects on technology performance in containing a contaminant plume. The simulations showed that the HFTW technology has the potential to contain the plume, although contaminant levels remained above MCLs. The concentrations above MCLs downgradient of the HFTWs in screened layers can partially be attributed to dispersion around the capture zone of the HFTWs. Additional HFTWs could be added to manage this problem. The potential for contaminant to bypass the treatment wells and travel to downgradient receptors, due to the existence of high conductivity zones that are not within the screened intervals of the wells, is a significant problem.

5.2 Conclusions

The HFTW model developed for porous media is appropriate for application in a fractured system. The literature review suggested that the assumption of an EPM when dealing with a fractured rock system is appropriate in the absence of detailed site data so long as the scale of the system being modeled was sufficiently larger than the scale of fracturing.

Numerical modeling is required to simulate HFTW operation for the hydrogeologic conditions encountered in the ROK. The heterogeneity of ROK aquifers requires a numerical model to examine the effects of an HFTW installation on contaminant fate and transport. A numerical model allows use of multiple layers to represent varying hydraulic conductivities and anisotropy with depth, in order to simulate the effect of these conditions on contaminant transport and technology performance. The numerical model developed by Huang and Goltz (1998), using input data obtained from well logs at Osan AB, provided a general understanding of how an HFTW system could be applied in a fractured rock aquifer in the ROK.

Model analysis indicates HFTW technology may be appropriate for containing contaminant plumes under hydrogeologic conditions encountered in the ROK, though bypassing of the treatment wells may be problematic. Contaminant concentrations modeled downgradient of the HFTW system were not below MCLs, though perhaps optimization of system variables (well spacing, number of wells, pump rates, and screen interval) could reduce downgradient contaminant concentrations to meet regulatory goals.

A significant problem that must be addressed when designing an HFTW system for application in fractured media is the potential for the contaminant to bypass the treatment wells by transport through unscreened high conductivity layers (Figure 4.3) or around the capture zone of the system (Figure 4.14). This would result in high contaminant concentrations downgradient. Extensive site characterization and engineering design measures (such as the use of multiple rows of treatment wells, screened over different depths) may serve to reduce this potential.

HFTWs have the potential to be a cost effective containment technology. In the literature review it was shown that HFTWs have the potential to be an order of magnitude cheaper than other technologies currently in use (predominately pump-and-treat). Since the DoD's ability to remediate contaminated sites overseas is hindered by the availability of money, cheaper containment technologies have a smaller impact on operation and maintenance (O&M) funds that are used for other mission requirements and installation priorities.

5.3 Recommendations

Explore the use of groundwater transport models other than the EPM to simulate the fate and transport of contaminants in ROK aquifers. Laboratory and field analyses can be used to correlate hydraulic conductivity with parameters like fracture density, spacing, and connectivity. This information would enable alternate groundwater flow models discussed in Chapter 2 to be explored. Results from these analyses could be used to validate the assumption that an EPM is appropriate for the ROK, or that another model more accurately predicts contaminant transport.

Investigate the processes of natural attenuation in ROK aquifers. The research showed that natural attenuation of chlorinated contaminants would be expected to be slow and not significant in scenarios typical of the ROK. However, the model simulations indicated that it would take the contaminant on the order of 10,000 days to travel 80 meters. Thus, even slow natural attenuation processes might turn out to be important. Studies of natural attenuation processes that might be occurring in fractured systems similar to those encountered in the ROK might support application of monitored natural attenuation as a remediation strategy, especially if receptors are far from contaminant source zones.

Optimize the HFTW system for application in a fractured rock aquifer. This study was a preliminary look at how an HFTW system might be applied in a fractured rock system, under conditions that might be encountered at Osan AB, Korea. No attempt was made to design an "optimal" system that would achieve specific performance goals at minimal cost. However, to ascertain the economic and technical feasibility of applying HFTWs under the hydrogeologic and contamination conditions found in the ROK, a more complete design analysis, perhaps using optimization techniques, will be required. Model analysis of an optimized design could be used to help determine the cost of the containment system, as well as to see whether downgradient contaminant levels below regulatory standards could be achieved at specific sites in the ROK.

Conduct a pilot test of the HFTW system in the ROK. For model validation and to enhance understanding of how an HFTW system could be applied in fractured media, a pilot study would provide important data on technology performance. These data would be valuable in determining the applicability of the HFTW technology to the ROK, as well as to other fractured rock aquifers.

Appendix A: Well data from Osan AB

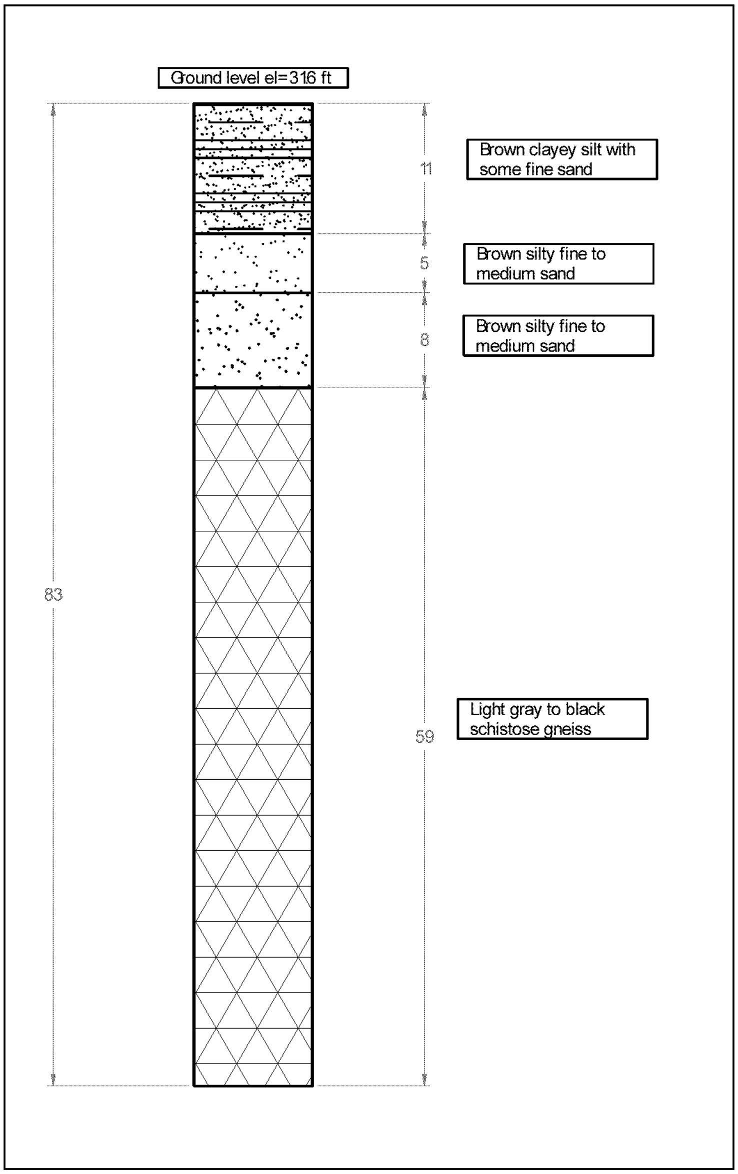


Figure A.1: well 9 at Osan AB, ROK

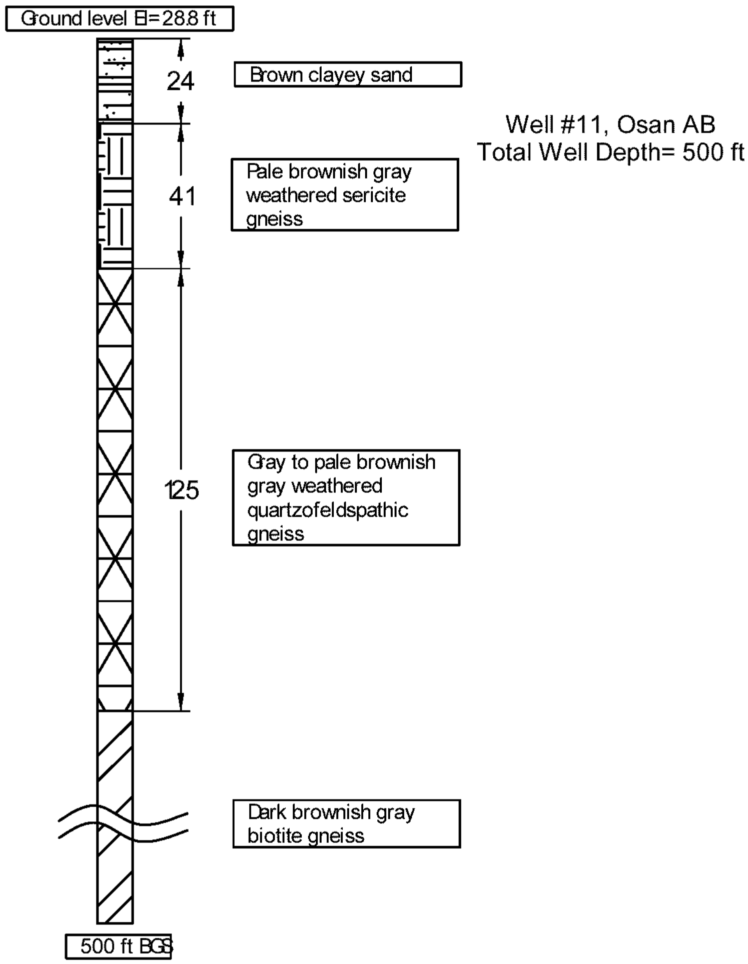


Figure A.2: well 10 at Osan AB, ROK

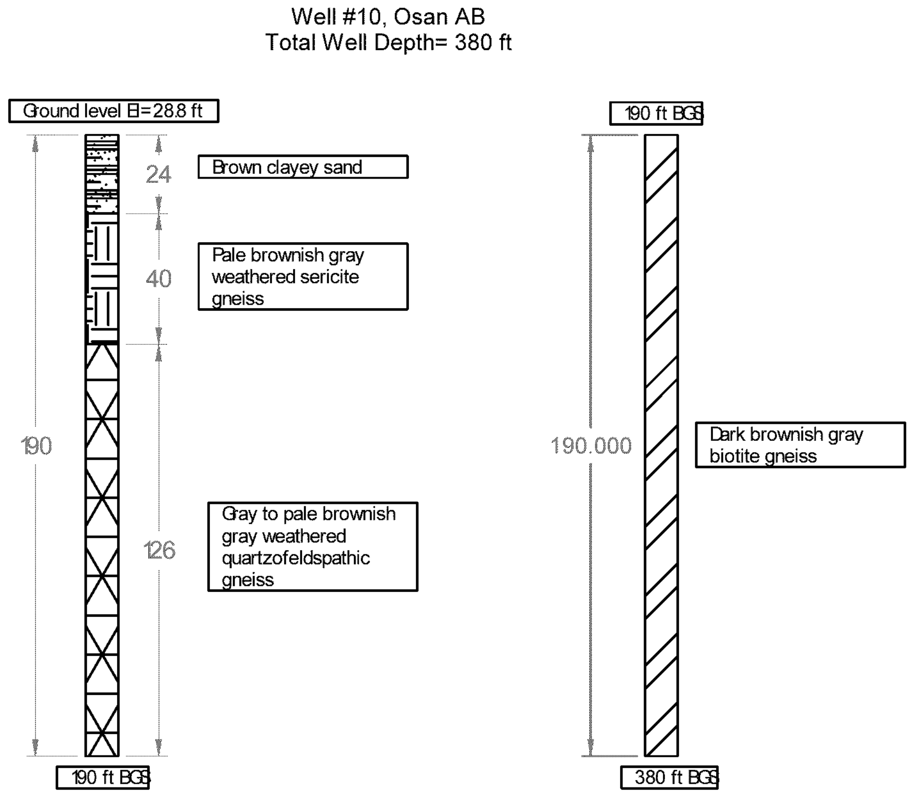


Figure A.3: well 11 at Osan AB, ROK

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REPORT DOCUMENTATION PAGE	*Form Approved* *OMB No. 074-0188*

1. REPORT DATE *(DD-MM-YYYY)*	**2. REPORT TYPE**	**3. DATES COVERED** *(From – To)*
26-03-2002	Master's Thesis	Sep 2000 – Mar 2002

4. TITLE AND SUBTITLE

AN ANALYSIS OF HORIZONTAL FLOW TREATMENT WELL APPLICABILITY FOR THE TREATMENT OF CHLORINATED SOLVENT CONTAMINATED GROUNDWATER AT UNITED STATES FORCES KOREA INSTALLATIONS

5a. CONTRACT NUMBER

5b. GRANT NUMBER

5c. PROGRAM ELEMENT NUMBER

6. AUTHOR(S)

Michael R. Staples, Capt, USAF

5d. PROJECT NUMBER
NA

5e. TASK NUMBER

5f. WORK UNIT NUMBER

7. PERFORMING ORGANIZATION NAMES(S) AND ADDRESS(S)

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AFIT/GEE/ENV/02M-14

9. SPONSORING/MONITORING AGENCY NAME(S) AND ADDRESS(ES)

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Brooks AFB TX 78235-5357 (210) 536-5661

10. SPONSOR/MONITOR'S ACRONYM(S)
AFCEE/ERT

11. SPONSOR/MONITOR'S REPORT NUMBER(S)

12. DISTRIBUTION/AVAILABILITY STATEMENT

APPROVED FOR PUBLIC RELEASE; DISTRIBUTION UNLIMITED.

13. SUPPLEMENTARY NOTES

14. ABSTRACT

Past research has shown that there is a rising public concern with environmental issues in the Republic of Korea (ROK). As Korean government and public interest in the environment grow, there is likely to be increased pressure to remediate environmental contamination at United States Department of Defense (DoD) installations in Korea. Impacting DoD's ability to remediate contaminated sites overseas is the fact that limited environmental funds must compete with high priority mission requirements. Thus, particularly at overseas bases, there is an urgent need for inexpensive and effective groundwater remediation technologies. This study focused on the containment of groundwater contaminated with chlorinated solvents in the fractured rock aquifers that are commonly encountered at DoD installations in the ROK. Horizontal Flow Treatment wells (HFTWs) were analyzed as a potentially cheaper, safer, and more effective technology for the containment of chlorinated solvent contaminated groundwater. Both hydrogeologic and design parameters were varied to determine their effects on the technology performance. From this study, it was determined that an HFTW numerical model developed for porous media is appropriate for application in the fractured systems encountered in the ROK, and that HFTWs have the potential to be a cost effective alternative for contaminant management in fractured media when compared to conventional technologies. The model analysis indicated HFTWs might be appropriate for containing contaminant plumes in the ROK, though bypassing of the treatment wells may be problematic.

15. SUBJECT TERMS

Groundwater contamination, Republic of Korea, fractured media, equivalent porous media, groundwater remediation, chlorinated solvents, groundwater modeling, remediation technology

16. SECURITY CLASSIFICATION OF:			**17. LIMITATION OF ABSTRACT**	**18. NUMBER OF PAGES**	**19a. NAME OF RESPONSIBLE PERSON**
a. REPORT	b. ABSTRACT	c. THIS PAGE			Professor Mark N. Goltz, ENV
					19b. TELEPHONE NUMBER *(Include area code)*
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Prescribed by ANSI Std. Z39-18

	Form Approved *OMB No. 074-0188*