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SIMULATION AND DESIGN OF PUMPING AND INJECTION SCHEMES
FOR AQUIFER RESTORATION UNDER VARIABLE HYDROGEOLOGIC
CONDITIONS

by

RICHARD L. SATKIN

A THESIS SUBMITTED
IN PARTIAL FULFILLMENT OF THE
REQUIREMENTS FOR THE DEGREE

MASTER OF SCIENCE

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ABSTRACT

SIMULATION AND DESIGN OF PUMPING AND INJECTION SCHEMES
FOR AQUIFER RESTORATION UNDER VARIABLE HYDROGEOLOGIC CONDITIONS

RICHARD L. SATKIN

The USGS MOC Model is a useful tool for evaluating different well patterns in an aquifer restoration scheme under variable hydrogeologic conditions. The best well pattern for a groundwater cleanup is highly site-specific and depends upon the objectives and constraints for each problem. In this research, seven different well patterns were studied to determine which well pattern(s) is the most efficient in achieving a range of desired levels of contaminant reduction. The well patterns were evaluated on the basis of cleanup time, volume of water circulated and volume of water requiring treatment. Eight generic hydrogeologic conditions were modeled using different combinations of drawdown, hydraulic gradient and dispersivity. The key hydrogeologic variables which control the rate of cleanup are well locations, pumping rates, transmissivity, dispersivity and hydraulic gradient. For a given set of well locations, by varying transmissivity and maintaining drawdown, dispersivity and hydraulic gradient constant, the cleanup time was found to be inversely related to the pumping rate.
I wish to extend special thanks to the following individuals whose assistance through the course of this study will always be greatly appreciated. My advisor, Dr. Philip B. Bedient, for his introduction to the problem, for the invaluable discussions and free exchange of ideas and his critical review of the manuscript. Drs. Mason Tomson and Clarence Miller for their review of the manuscript. Karen Miller for her critical review of the manuscript, her moral support and many helpful discussions. Scott Ziegenfuss, Chuck Newell, Hanadi Rifai and Ernesto Baca for their many helpful discussions. Jill Oglesby for her excellent drafting skills. Finally, my warmest thanks to my loving wife Phyllis Bryer and my daughter Dori whose unselfish support and encouragement made it all possible.
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1.0 INTRODUCTION

Groundwater contamination in the U.S. is widespread today. Because of the poor disposal practices used over the last fifty years to rid the immense volumes of wastes generated, and as a result of the proliferation of chemicals that have sustained our industrial society, our aquifers are now polluted. Fortunately, recent state and federal legislation has mandated the prevention of further groundwater pollution and the costly cleanup or restoration of our national water supply.

The question of whether a polluted aquifer can be restored and how best to proceed is dependent upon the hydrogeologic and geochemical properties of the aquifer and on the chemical and physical properties of the contaminant (Konikow and Thompson, 1984). Common restoration options typically considered include the following: 1) the no action alternative which relies on natural attenuation and dilution after source removal; 2) groundwater pumping; 3) containment, either physical and/or hydraulic; 4) excavation and removal of the contaminated part of the aquifer; and 5) in situ biological or chemical treatment.

Groundwater pumping is probably the most common method employed for aquifer restoration. Groundwater pumping lowers the watertable and removes the contaminated water from the aquifer. Water is pumped by one well or multiple wells to the surface for removal or on-site treatment and disposal or reinjection. Reinjection of treated water eliminates the potentially high cost of disposal and at the same time accelerates the removal of contaminants by increasing the natural gradient. Reinjection can also be used to create a hydraulic barrier to prevent further plume migration.
Pumping and injection wells are often an integral part of an overall aquifer restoration scheme. In addition to providing hydrodynamic control to contain the contaminant plume to a specified area, the wells may play a more active role by delivering and removing chemical substances for an in situ treatment process or removing the contaminant for surface treatment and reinjection. The objective in a pumping and injection scheme lies with the creation of a groundwater divide or a capture zone which completely encompasses the contaminant plume. The key variables which control the efficiency of the withdrawal scheme and the size and shape of the capture zone depend upon the contaminant chemistry, well locations, pumping rates and the aquifer properties such as transmissivity, dispersivity and hydraulic gradient.

This research focuses on the evaluation of different well patterns used for a withdrawal scheme under variable hydrogeologic conditions. Seven different well patterns were studied to determine which well pattern(s) is the most efficient in achieving a range of desired levels of contaminant reduction. The well patterns were evaluated on the basis of cleanup time, volume of water circulated and volume of water requiring treatment. Eight generic hydrogeologic conditions were modeled using different combinations of drawdown, hydraulic gradient and dispersivity. Finally, different well patterns were investigated for application to an actual industrial waste site.
2.0 PREVIOUS WORK

Analytical and numerical modeling are both commonly used to screen and evaluate different remedial options for aquifer restoration. Depending upon the desired level of accuracy, complexity of the site, and the data and resource availability, will dictate whether a simple analytical solution or a complex numerical method is required. Analytical methods are restricted to highly idealized groundwater flow conditions where the velocity is steady and uniform, or radial, as in flow to wells. The aquifer is assumed to be homogeneous and isotropic. Analytical methods may be simple to use, but may not be applicable to a field site with complex boundary conditions.

Numerical models are more flexible than analytical solutions because complex boundary conditions with various combinations of pumping and injection wells can easily be approximated by arrangement of grid cells. Numerical techniques commonly employed for simulating groundwater transport include finite difference and finite element methods and the method of characteristics (MOC). The method of characteristics (MOC) is most useful when solute transport is dominated by convective transport (Konikow and Bredehoeft, 1978), as in the case of most aquifer restoration schemes involving pumping and injection. The method of characteristics (MOC) also minimizes numerical dispersion, a problem plaguing most numerical methods.

To select the best well configuration for a particular withdrawal scheme requires the ability to predict changes in flow and chemical concentration in the aquifer for each possible management alternative (Konikow and Thompson, 1984). A numerical model with ground water solute transport such as the USGS MOC Solute Transport Model (Konikow and Bredehoeft, 1978) is well suited for
this purpose. The best pumping arrangement is developed generally by a tedious trial and error process (Glover, 1982). The trial and error approach suffers because it is inefficient; however, the heuristic knowledge gained by the user is invaluable and allows the modeler to steadily improve on future trials.

Many investigators have used numerical ground water models as a tool in the design of aquifer restoration strategies because they provide a rapid means of predicting or assessing the effects of different remedial alternatives. Andersen et al., (1984) used a finite difference ground water model as an aid in selecting an appropriate remedial action at the Lipari Landfill in New Jersey. They made 14 computer simulations to test a slurry wall, drain location, and a clay cap. A slurry wall was simulated in the model by using a very low hydraulic conductivity. Different locations for a drain were simulated by treating the finite difference block containing the drain as a constant-head node. A clay cap was simulated by setting the recharge in the finite difference blocks representing the cap equal to zero. Pendrell and Zeltinger (1983) used a 2-D finite difference model to evaluate alternative remediation strategies at the Rocky Mountain Arsenal.

Althoff et al., (1981) used a ground water flow model to test a variety of well configurations, well locations, and pumping rates for hydraulic capture of a 1000 ft long 1,1,1-trichloroethane plume. In order to assess optimum pumping rates for a hydrodynamic isolation system consisting of pumping wells and an infiltration gallery, Ozbilgin and Powers (1984) used both 2-D and 3-D computer models. Bedient and Baca (1986) made 15 computer simulations using the USGS MOC Model for a Remedial Investigation/Feasibility Study (RI/FS) at a Superfund site at Conroe, Texas. They evaluated a number of restoration schemes including a slurry wall barrier and both the five-spot and nine-spot well patterns. Freeberg et al., (1987) used the USGS MOC Model to delineate a trichloroethylene plume and
to evaluate different withdrawal schemes at an industrial waste site.

Using a finite element model, Tsai and Zielen (1985) compared six pumping schemes for withdrawal of groundwater contaminated with explosive wastes, including 2,4,6-trinitrotoluene (TNT), at an ammunition plant in Nebraska. They found that extraction wells located along the plume axis are more effective than pumping schemes with the same number of wells located in groups over the plume area. Gray and Hoffman (1983) used a 2-D finite element model to investigate two remedial pump / treat and reinjection schemes for a 4000 ft long plume at Price's Landfill in New Jersey. At the Love Canal site in New York, Cohen and Mercer (1984) evaluated the hydraulic benefits of a synthetic cover, French drain, and concrete cutoff wall using a 2-D finite element model. They concluded that the benefits of the proposed barrier wall were minimal compared to the proposed synthetic cover.

The trial and error approach of investigating different remedial strategies involving potential well locations and pumping rates with a contaminant transport model is very tedious and the method does not guarantee that the best solution will be found. Gorelick of the USGS along with others in a series of papers (1983, 1984, 1986) have demonstrated how linear or nonlinear programming (optimization techniques) can be combined with a ground water transport model to efficiently arrive at an optimal design strategy. Simulation-management models, as they are often called, are formulated with an objective function subject to a series of constraints. The objective function could be to minimize the amount of time to achieve a desired level of cleanup in the aquifer. The governing equations which describe solute transport are the constraints. The constraints may be formulated by the user so as to control groundwater heads, gradients, velocities or pumping. Molz and Bell (1978) used linear programming
to determine the pumping rates for four wells that would create a stagnation zone or zero hydraulic gradient in an area to be used for fluid storage. Shafer (1984) demonstrated how nonlinear optimization techniques coupled to a ground water transport model can be used to determine pumping rates to: 1) create a stagnation zone; 2) create hydraulic barriers to pollutant migration; 3) control or steer the trajectory of a plume; and 4) intercept the trajectory of a plume. An optimization technique is not without limitations. For some highly nonlinear problems convergence to an optimum solution cannot be guaranteed, and the rate of convergence may be slow (Shafer, 1984).

Semianalytical models, although less complex than numerical codes and optimization techniques, are advantageous because they are easy to use. The semianalytical model Ressq (Javandal et al., 1984) is based upon the complex velocity potential theory. Ressq is a powerful tool because of its computer generated plots of groundwater streamlines. It can be used to evaluate how the removal capacity of a pumping system is affected by different well configurations, pumping rates and hydraulic gradients. Javandel and Tsang (1986) present type curves based upon complex velocity potential for determining the optimum number of pumping wells, their rates of discharge, and locations for creating a capture zone using one to three pumping wells.

Analytical equations are available in the literature (Bear, 1979) which address solute transport in a uniform flow field under simplified boundary conditions. Few of these however, account for dispersive mixing and the presence of multiple pumping and injection wells. Analytical equations which solve for hydraulic head in a homogeneous aquifer are also available for most of the common well patterns, but none of these address solute transport. Muskat (1937) provides a solution for estimating the direct travel time of a particle of water
from an injection well to a production well: \( t = \frac{4\pi nd^2b}{3Q} \). Wilson (1984) presents equations and dimensionless graphs for estimating hydraulic travel times and capture zones for both the doublet and double-cell well patterns. The double-cell well pattern consists of two pumping wells and two injection wells oriented to create an inner and outer recirculation cells. The outer cell reduces the flushing time and the amount of water requiring treatment, and also provides a back-up if the inner cell should fail. See Section 5.8 for a further discussion of the double-cell well pattern.

Gillham (1982) applied simple 1-D analytical transport models to estimate the total volume of water that must be withdrawn via a pumping well in order to evaluate the feasibility of aquifer restoration. A simple advection model predicts that the volume of water that must be removed is equal to the volume of contaminated water initially present. With the advection-dispersion model, more water has to be removed than would be predicted on the basis of the advection model. For reactive species undergoing retardation, the volume of water that must be removed is dependent on the magnitude of \( K_d \), the partition coefficient of the species between the solution and solid phase. Only when the value of \( K_d \) is small would aquifer restoration by water removal be feasible. For these types of analyses, highly idealized boundary conditions are assumed and include a uniform velocity and uniform concentration everywhere in the aquifer. For this research, a 2-D numerical model, which is considerably more versatile than a 1-D analytical model, was used.
3.0 NUMERICAL MODEL

The USGS MOC Solute Transport Model (Konikow and Bredehoeft, 1978) is one of the most widely used 2-D groundwater transport models. It can simulate transport in one or two dimensions under steady state or transient flow. The model computes changes in concentration over time caused by the processes of convective transport, hydrodynamic dispersion, mixing (or dilution) from fluid sources, first-order decay and equilibrium sorption-desorption (Konikow and Bredehoeft, 1978). The model is well suited for this research because it is well documented, has simple input/output formats that can easily be modified, allows any number and arbitrary placement of injection and pumping wells, and has time variable pumping periods.

To simulate solute transport, the computer program solves two partial differential equations simultaneously using numerical methods. It uses an iterative alternating direction implicit (ADI) procedure to solve a finite difference approximation to the groundwater flow equation, and it uses the method of characteristics (MOC) to solve the solute transport equation (Bedient et al., 1985). In the method of characteristics, advective solute transport is simulated by placing a number of particles in each cell of the finite difference grid and tracing the movement of these particles through time. Changes in concentration caused by hydrodynamic dispersion, fluid sources and changes in saturated thickness are calculated using an explicit finite difference approximation to the dispersion equation. Konikow and Bredehoeft (1978) and Bedient et al., (1985) give a more detailed description and mathematical development of the model.

Two major factors which affects both the accuracy of the numerical solution, and comparison with the analytical solution, are the time step and size of the
grid cell. Selection of too large a grid cell, would require many moves for the tracer particles to complete a given time step. This results in very long computation time and associated cost, and can affect the comparison with the analytical solution as discussed below. One of the problems encountered in this study with the MOC particle tracking technique to compute solute transport is illustrated in Figure 3.1. During an aquifer cleanup, portions of the flow field are strongly radial and advection dominated. Although the contaminant plume is initially contained within the capture zone, particles can be convected across the groundwater divide outside the capture zone. This is due to the linear paths which the particles take to approximate the curvilinear flow field. This problem can be minimized by decreasing the time step and size of the grid cell.

Dispersion is one of the two major processes governing solute transport. It is a measure of the spreading and dilution of a contaminant. Up until recently, dispersivity values for contaminant transport models have been used as a calibration parameter, being adjusted until the model correctly reproduces the observed concentration distribution (Anderson, 1984). These values in the range of 3 to 200 m may not be physically meaningful (Anderson, 1984). Recent theoretical and field studies (Gelhar et al., 1983; Molz et al., 1986; Freyberg, 1986; LeBlanc and Hess, 1987), have shown that physically realistic dispersivity values approaching those measured in the laboratory (10^{-2} to 1 cm) can be obtained if the statistical and detailed vertical hydraulic conductivity distributions are known. In 2-D models, where vertical hydraulic conductivity profiles are not available, the modeler inevitably ends up adjusting the dispersivity during the calibration process (Molz et al., 1986). See Section 5.5 for a further discussion on dispersion.
Figure 3.1 Part of hypothetical finite difference grid showing how contaminant initially within the capture zone may be advected outside the capture zone by the particle tracking method.
The computer model represents the flow field using a rectangular, block-centered finite difference grid (Figure 3.2). In this study a 15 x 25 cell grid was used with each cell representing a 50 x 50 ft area. Two rows of constant head cells were specified to produce the regional flow field. Transmissivity and thickness were set to constant values and recharge was set to zero. The storage coefficient was set equal to zero in order to simulate a steady state flow. In many contaminant transport problems, the velocity and hydraulic head distributions can be considered to be invariant with time i.e., a steady state flow field exists. The steady state flow assumption is valid because the velocity field is readily established compared to the typically long lengths of time required for a cleanup.
Figure 3.2 Grid and boundary conditions for aquifer model.
4.0 OBJECTIVES

The objective of this research is to numerically simulate the removal of a non-reactive contaminant plume using different well configurations for several generic hydrogeologic conditions. The results of this research will aid decision makers in the selection process of determining not only if the withdrawal option is feasible but also how to design the most cost-effective pumping and injection scheme. The knowledge gained from this study can also be applied to such nonconventional treatment technologies as in situ bioreclamation and chemical treatment. These in situ strategies commonly require the recirculation of surfactants, nutrients or other chemicals during the restoration process. Well pattern spacing and pumping rates control the recirculation and must be designed to ensure that the chemicals make proper contact with the pollutant and have a sufficient residence time before being recycled again. After treatment, degradation products may have to be recovered (Wagner and Kosin, 1985). Proper placement of wells is crucial for an in situ treatment method to be successful.

In order to determine what is the best or optimum location for well placement, the objectives of the restoration strategy first need to be clearly defined. The objective may be to prevent contamination of a major water supply well and remove the pollutant plume while incurring the least expense. Oftentimes, the objectives are constrained by site-specific physical, chemical, economic and political criteria. For example, the saturated thickness of an aquifer limits the amount of drawdown, thereby constraining the maximum pumping rate. Costs may limit the number of wells or the type and sizing of the treatment facility. In turn, the volume or maximum concentration of pollutant to be treated might have to be maintained below a certain level. Prevention of the pollutant from moving off a property boundary may be another constraint.
In this study, the objective was to reduce the maximum contaminant concentration in the plume by a range of levels, up to 99.995%, as quickly as possible, and at the same time not allow any contaminant to migrate further than fifty feet from the downgradient edge of the plume. The pumping rates were constrained by drawdown. Results are presented for both a high and low series of drawdowns corresponding to \( \leq 10 \text{ ft} \) and \( \leq 5 \text{ ft} \), respectively. Injection rates were set equal to pumping rates so that no water from an external source would be required. In well patterns involving no injection, water disposal would be required. The procedure is discussed in Section 7.0.
5.0 METHODOLOGY

5.1 Generic Characterization

As initially conceived, the objective of this project was to evaluate the effectiveness of different well patterns under several different generic hydrogeologic conditions. In order for the results of this project to have widespread applicability, values for the groundwater model parameters which describe the pollutant transport and withdrawal scheme needed to be selected which are representative of actual field sites. The key variables are hydraulic conductivity, aquifer thickness, hydraulic gradient, dispersivity, drawdown, concentration, plume shape and size and well pattern.

5.2 Hydraulic Conductivity

Hydraulic conductivity, K, is perhaps the most difficult parameter to assign at a hazardous waste site without a very detailed sampling program. In groundwater modeling, hydraulic conductivity is normally the most sensitive parameter. One of the reasons is that K can vary over a large range for a typical site (approximately 10 orders of magnitude). Hydraulic conductivity data often exhibits a large spatial variability with no apparent correlation even among closely spaced samples. This is not surprising when viewed in light of the nature of the depositional processes of sedimentary rocks. The literature reports that the hydraulic conductivity of a formation tends toward a lognormal distribution (Freeze, 1975; Freeze and Cherry, 1979). Hydraulic conductivity is embedded in the transmissivity term as required input for the USGS MOC Model. The ratio of $\frac{T_{xx}}{T_{yy}}$ allows specification of anisotropy. However, for the sake of applicability and ease of use, transmissivity is considered homogeneous and isotropic for this research. Because of the wide range of values possible for K it is impossible to
assign a generic value. Hydraulic conductivity values of $10^{-5}$, $10^{-4}$, $10^{-3}$, $10^{-2}$, and $10^{-1}$ ft/sec (multiply by 30.48 to obtain cm/sec) were initially selected for the study, which range from a typically fast gravel aquifer to a slower, silty sand aquifer.

5.3 Aquifer Thickness

The aquifer thickness, B, is a highly site-specific parameter and is dependent upon the local stratigraphy and geologic structure. In contaminant transport, the aquifer thickness influences the opportunity for vertical dispersive mixing as the contaminant plume moves downgradient. Furthermore, the extent of the aquifer subject to contamination may be significantly less than the aquifer thickness, particularly where there are immiscible contaminants or large density differences. In the USGS MOC Model, vertical variations in concentration and piezometric head are assumed to be negligible. During aquifer restoration, the maximum amount that a well can pump is constrained by the thickness of the aquifer; drawdown cannot exceed the thickness of the saturated zone. In this study, the aquifer thickness was set to a constant value of 10 ft. Ten feet represents a fairly typical value. The transmissivity term may be adjusted for applications to sites with different values of aquifer thickness.

5.4 Hydraulic Gradient

The hydraulic gradient is roughly equivalent in magnitude and direction to the topographic slope at a regional scale. At the scale of a site investigation, however, the hydraulic gradient may differ substantially due to man-made influences, including pumping wells and areas of enhanced recharge (landfills) (Guswa et al., 1984). The hydraulic gradient presently measured at an abandoned waste site where disposal ponds may have once been operative is typically lower
than such measurements made in the past when there may have been groundwater mounding. For this research, cleanup simulations were made with both a low (.0008) and a high (.008) background piezometric gradient. The regional flow regime was incorporated into the model as two rows of parallel constant head boundaries at both ends of the 15 x 25 finite difference grid used in the analysis.

5.5 Dispersivity

The dispersion coefficient, $D$, is an important parameter because it controls the degree of spreading and dilution of a pollutant plume. The dispersion process results from molecular diffusion and mechanical mixing. Mechanical mixing is attributed to both hydrodynamic dispersion, resulting from velocity variations in each pore channel, and macrodispersion, resulting from small scale velocity variations due to variations in hydraulic conductivity (Freeze and Cherry, 1979). The dispersion coefficient is made up of a molecular diffusion component and a dispersion component:

\[ D_l = a_1 V + D^* \]  \hspace{1cm} \text{Eq 5.1}

where

- $D_l$ = longitudinal dispersion coefficient
- $a_1$ = longitudinal dispersivity
- $V$ = average seepage velocity
- $D^*$ = molecular diffusion coefficient

The hydrodynamic dispersion process becomes less important at the field-scale where the macrodispersion process tends to dominate. Since the porous media are not homogeneous, this results in solute fingering through higher permeability layers. Mechanical mixing is the dominant process at the relatively high velocities typically encountered in aquifer restoration schemes. The molecular
diffusion component becomes significant only when groundwater velocities are very slow. In the USGS MOC Model, the molecular diffusion term is neglected.

Several investigators (Anderson, 1979, 1984; Pickens and Grisak, 1981) have noted that dispersivity varies with the scale of the analysis (Figure 5.1). Laboratory values range from $10^{-2}$ to 1 cm, while field studies range from 3 to 200 m (Anderson, 1979). Pickens and Grisak (1981) reported a simple linear dependency for longitudinal dispersivity, $a_1$, as:

$$a_1 = 0.1 \times x$$  \hspace{1cm} \text{Eq 5.2}

where $x$ = mean travel distance

Until very recently, it was not clear whether the dispersivity increases indefinitely with scale or reaches an asymptotic value as predicted by recent stochastic theories (Gelhar et al., 1985). Data obtained from a contaminant plume over two miles in length at the Otis Air Force Base at Cape Cod, Massachusetts, indicates that the longitudinal dispersivity, at this site, approaches an asymptotic value of 1 m. The transverse dispersivity is 2\% (0.02 m) of the longitudinal dispersivity (LeBlanc and Hess, 1987). At the Borden landfill in Canada, the asymptotic longitudinal dispersivity after 1038 days of travel, approaches 0.5 m, however, asymptotic conditions were apparently not reached. The transverse dispersivity is 10\% (0.05 m) of the longitudinal dispersivity and shows no increase in magnitude, analogous to the behavior of the longitudinal dispersivity (Freyberg, 1986).

In order to obtain an accurate measure of field-scale dispersion, the hydraulic conductivity, hydraulic head and porosity distributions need to be known. Because of the difficulty in obtaining such a detailed measure of the spatial variability of the aquifer and the long travel times associated with a contaminant plume it is very difficult to obtain a good field measure of dispersivity. Gelhar et al (1985)
Figure 5.1 Longitudinal dispersivity versus scale of observation, after Gelhar et al., 1985.
provides an excellent critique of 55 reported field-scale dispersivity tests. The best study to date, is an ongoing, natural gradient field tracer experiment, using multilevel sampling, and is being conducted at the Borden landfill site in Canada (Freyberg, 1986). Transverse and vertical dispersivity have been studied to a lesser degree than longitudinal dispersivity. In this study both a high longitudinal dispersivity (30 ft) and a low longitudinal dispersivity (10 ft) were used in the simulations. In the USGS MOC Model, dispersivity is constant both temporally and spatially. A transverse dispersivity of 30% of the longitudinal dispersivity was used in the model and may be high based upon recent field studies.
5.6 Drawdown

In designing an aquifer cleanup operation, the selection of the number of pumping wells, discharge rates and well locations is typically constrained by the maximum allowable drawdown, which is governed by the saturated thickness. As many contaminated aquifers are located at shallow depths, the maximum allowable drawdown at the well should reflect these situations. In the model simulations, two sets of maximum allowable drawdown were used, a large drawdown ($\leq 10$ ft) and a small drawdown ($\leq 5$ ft).

When one assigns a discharge rate to a well in the simulation model, the discharge is apportioned equally throughout that 50 x 50 ft cell representing the pumping well in the finite difference grid. The computed model drawdown for the cell represents an average drawdown over the entire 2500 sq ft cell and is not the drawdown in a well. Steady state drawdown from the model output was compared to the drawdown predicted by the Thiem equation:

$$s = \frac{Q}{2\pi T} \ln \frac{r_1}{r_2}$$  \hspace{1cm} \text{Eq 5.3}

In this comparison, $r_1$ and $r_2$ are set equal to the radial distances between each grid cell and the cell containing the well. It was observed that the model always over-predicted the drawdown near the well by approximately 15% compared to that predicted by the Thiem equation. However, near the constant head boundaries, the model will always under-predict the drawdown because of the fixed head (zero drawdown) at the boundary. The reason for the differences between the drawdowns predicted by the model and the Thiem equation is attributed to the different boundary conditions. The analytical solution assumes an infinite, unbounded aquifer with a horizontal piezometric surface, whereas the
modeled aquifer is finite. Figure 5.2 illustrates that for a given discharge rate, the cone of depression is steeper in a bounded aquifer than in an unbounded aquifer as there is less water to draw from in the bounded case.
Figure 5.2 Effect of different boundary conditions on the cone of depression.
5.7 Initial Plume and Cleanup Criteria

The contaminant plume used in this study is shown in Figure 5.3. The maximum dimensions of the plume are 650 x 350 ft. It has a length to width ratio of approximately 3:1 over the area with significant concentrations. The overall size and shape of this plume is representative of several others at field sites investigated by Rice University workers. Many of the hydrocarbon plumes studied are characterized by a long and narrow shape. The small amount of lateral spread has been partially attributed to biodegradation at the edges of hydrocarbon plumes (Borden and Bedient, 1986).

The initial contaminant plume used in this study was created using an injection well. Adjustments of any source loading or aquifer parameters were found to have a significant effect on the concentration, shape and extent of migration of the plume. Table 5.1 summarizes the aquifer characteristics and source loading parameters used in creating the plume with the USGS MOC Model. The objective in creating this plume was not to match a known plume, but rather to create a plume having a typical tear-drop shape with a maximum concentration of 1000 units. This concentration has no specific units. For application to other sites, the source loading and aquifer characteristics used in creating the plume are irrelevant. It is important, however, to have a plume with a similar size and shape. Results are presented with dimensionless concentration for widespread application.

The simulations were carried out until the maximum concentration everywhere in the aquifer was < 0.05 units. This shows up as 0.0 units in the
Figure 5.3 Contaminant plume used in this study.
<table>
<thead>
<tr>
<th>Parameter</th>
<th>Value</th>
</tr>
</thead>
<tbody>
<tr>
<td>Well location</td>
<td>col = 8, row = 10</td>
</tr>
<tr>
<td>Injection rate</td>
<td>0.25E-5 ft²/sec (1.6 gpd)</td>
</tr>
<tr>
<td>Concentration *</td>
<td>15000 units</td>
</tr>
<tr>
<td>Pumping period</td>
<td>30 yr</td>
</tr>
<tr>
<td>Hydraulic gradient, i</td>
<td>0.0008</td>
</tr>
<tr>
<td>Hydraulic conductivity, K</td>
<td>0.0001 ft/sec</td>
</tr>
<tr>
<td>Thickness, b</td>
<td>10 ft</td>
</tr>
<tr>
<td>Longitudinal dispersivity, a_l</td>
<td>10 ft</td>
</tr>
<tr>
<td>Transverse dispersivity, a_v</td>
<td>3 ft</td>
</tr>
<tr>
<td>Effective porosity, n</td>
<td>30%</td>
</tr>
</tbody>
</table>

* Concentration can be any set of units.

Table 5.1 Source loading and aquifer characteristics for creating the initial plume.
observation well hydrograph output routine of the model. This gives a relative concentration of \(0.05/1000. = 0.00005\) which represents a 99.995\% concentration reduction.
5.8 Well Patterns

There are many possible arrays or patterns in which to place groundwater wells. The seven different well patterns, shown in Figures 5.4a and 5.4b, were evaluated for their effectiveness in a remedial cleanup. These well patterns include the standard single, doublet, 3-spot, and 5-spot patterns in addition to the centerline, double-cell (after Wilson, 1984) and double-triangle patterns. The direct line-drive well pattern consists of a row of injection wells upgradient and a row of production wells downgradient. This pattern was eliminated from the study because for this size and shape contaminant plume, it required placing wells in adjacent grid cells. Although mass balance errors observed for the line-drive pattern were < 8 %, numerical instabilities in the model may occur when sources or sinks are placed in adjacent grid cells (Mary Wheeler, personal communication, 1986).

The simplest case is a single extraction well located somewhere within the plume boundary. The centerline consists of three extraction wells separated by one or two grid cells located along the axis of the plume in the direction of flow. The doublet consists of an injection well and a production well pair. The injection well is typically placed upgradient. Under ideal conditions, the production well will only be pumping water that has previously been cycled through the treatment system and injection well. The doublet, as in any production/injection scheme improves the flushing rate over that of a single production well because of the increased hydraulic gradient towards the production well. However, the doublet also leads to larger volumes of water requiring treatment with smaller concentrations. The maximum pumping rate in the doublet is greater than that of a single well because of the superposition of the groundwater mound produced by the recharge well upon the cone of depression.
Figure 5.4a Well patterns used in this study.
Figure 5.4b Well patterns used in this study.
The 3-spot pattern consists of recharge wells both upgradient and downgradient of the plume with an extraction well between the pair. Under high gradients, the 3-spot pattern may be superior to that of a doublet because of the hydraulic barrier created by the downgradient injection well. The downgradient recharge well could also provide a backup to the production well if the well should fail. The 5-spot well pattern is commonly used in the petroleum industry particularly in secondary recovery by water flooding. The 5-spot consists of an extraction well surrounded by four injection wells.

The double-cell hydraulic containment system (Wilson, 1984) consists of an inner cell and an outer recirculation cell using four wells along a line bisecting the plume. The inner cell can be designed large enough to just capture the plume. The outer cell serves to create a steep hydraulic gradient through the plume by circulating a much larger volume of water than the inner cell. The presence of the outer cell reduces the time required to capture the plume and the amount of water requiring treatment. This well pattern has several advantages over the single cell (doublet) pattern: 1) volume of water requiring treatment is minimized and pollutant concentration is maximized; 2) flushing time is reduced; and 3) the plume is contained and the outer cell serves as a backup should the inner cell fail. Site access and availability for placement of the outer cell wells may pose a potential problem for the use of a double-cell well pattern. Also, in some cases, a significant drawdown may be required in order to obtain the necessary volume of water for the outer cell design.

The double triangle consists of six wells with three wells located upgradient and three wells downgradient. This pattern represents a modification of the double-cell hydraulic containment system.
In both the single and centerline well patterns, there is no injection. In the other well patterns, the injection rate was set equal to the pumping rate so that the total volume of water withdrawn from the aquifer was reinjected back into the aquifer. In this research, the volume of water injected was always equally distributed among each injection well.
6.0 ANALYTICAL METHODS

Two prime objectives of aquifer restoration are to contain and/or remove pollutant plumes (Wilson, 1984). Using injection and extraction wells is a common technique to accomplish both of these objectives. The concept behind a withdrawal scheme lies in the creation of a groundwater divide or capture zone. The capture zone is defined to be that area of an aquifer that actually yields water to a pumping well. A capture zone for a single pumping well in a uniform flow field is shown in Figure 6.1. A capture zone is created by inducing changes in the local flow rate and hydraulic gradient.

Keely and Tsang (1983) have shown that the capture zone is generally much smaller than and does not coincide with the cone of depression due to pressure influence. With moderate hydraulic gradients, a flow line may pass between two adjacent wells even though their cones of depression overlap. Only under idealized stagnant aquifer conditions (no background hydraulic gradient) would a capture zone coincide with the cone of depression.

The principle of superposition is a useful technique that allows the analytical solution of a complicated problem to be obtained through the linear combination of a number of elementary solutions. Superposition is valid for homogeneous linear partial differential equations. A common use of the principle of superposition is in the calculation of steady state drawdown for a multiple well system in confined and semiconfined aquifers. The resultant drawdown at any point in the aquifer can easily be obtained by summing the drawdowns or buildups induced by each well individually. For unconfined aquifers, superposition or linearization can be employed when variations in the hydraulic head, \( h \), are small and an average \( h \) can be used to describe the flow. The analytical expressions
Figure 6.1 Capture zone for a single well in a uniform flow field, after Bear, 1979.
describing the stagnation point and the groundwater divide are both developed below using superposition, by combining the effect on the flow field due to the ambient groundwater flow with the effect on the flow field due to pumping.

The equation for a capture zone can be derived by evaluating the stream function, \( \psi \), for a single pumping well in a uniform flow field:

\[
\psi = \frac{q_o b y}{T} + \frac{Q_w}{2 \pi T} \tan^{-1} \left( \frac{y}{x} \right) \tag{Eq. 6.1}
\]

where

- \( q_o \) = specific discharge
- \( b \) = aquifer thickness
- \( T \) = transmissivity
- \( Q_w \) = pumping rate

Multiplying both sides by \( 2 \pi T/Q_w \), taking the tangent, and setting \( \psi = 0 \), an arbitrary constant, yields the equation which describes the capture zone or groundwater divide for a single well:

\[
y/x = \pm \tan \left( 2 \pi q_o b y/Q_w \right) \tag{Eq. 6.2}
\]

+ for \( y > 0 \)
- for \( y < 0 \)

The stagnation point, \( S \), is the point at some distance downgradient of the pumping well where the pull of water back toward the well is exactly countered by the natural flow in the aquifer. Thus, the stagnation point is the point at which the resultant velocity or \( d \phi/dx \) (where \( \phi \) is the velocity potential) is equal to zero. For a single pumping well in a regional flow field parallel to the \(-x\) direction, the velocity potential, \( \phi \), is defined as

\[
\phi = \frac{q_o b x}{T} + \frac{Q_w}{4 \pi T} \ln(x^2 + y^2) \tag{Eq. 6.3}
\]
The stagnation point \((x=x_s, \ y=0)\) is found by taking \(d\phi/dx\) and setting it equal to zero. For a single well, the stagnation point occurs at the point whose coordinates are:

\[
x_s = -\frac{Q_w}{2\pi b q_o}, \ y = 0
\]

Eq 6.4

The maximum width of the upgradient inflow zone is equal to \(2\pi\) times the stagnation distance.

DaCosta and Bennet (1960) give equations which describe the locations of stagnation points for a recharging and discharging pair of wells (doublet) that are aligned at various angles to the direction of areal flow. Using the equations developed by DaCosta and Bennet (1960), Grove (1970) extended their analysis to allow the calculation of the times of arrival and concentrations of the injected fluid along various streamlines between a recharging and discharging well pair.

Wilson (1984) presents expressions and dimensionless graphs for the maximum width of a capture zone for both the doublet and double-cell well patterns where the ambient flow is parallel to the pair(s) of wells. The expression for the maximum width of a capture zone for a doublet is:

\[
1 = Tq_w/Q_w + 2\pi \tan^{-1}(w/2a)
\]

Eq 6.5

where

\[
w = \text{width of contaminant zone}
\]

\[
2a = \text{distance between wells}
\]

Assumptions inherent in the analytical equations given above include horizontal flow in a homogeneous and isotropic aquifer and fully penetrating wells. It can be shown that for a given pumping rate, a partially penetrating well could capture a wider plume in the horizontal plane than a fully penetrating well.
(Ozbilgin and Powers, 1984). The gain in the horizontal plane is accompanied by a loss in the vertical plane. Another factor influencing the size of the capture zone not considered in the analytical expressions is recharge from precipitation, which will act to decrease the size of the capture zone.
7.0 PROCEDURE

The initial step in making the simulations was to select the hydrogeologic parameters under which the different well patterns were to be tested. A high value or a low value for each of drawdown, hydraulic gradient, and longitudinal dispersivity was selected. The first set of runs investigated had a drawdown of $\leq 10$ ft (high value), a hydraulic gradient of 0.0008 (low value), and a dispersivity of 10 ft (low value). A hydraulic conductivity, $K$, of $10^{-4}$ ft/sec was used in this first set of runs.

The single well was always the first pattern investigated when any hydrogeologic parameter was changed. The pumping rate was estimated by using the Thiem equation. The approximate length of time required for the simulation was determined using the seepage velocity. In order to narrow down the possible locations for the well, the capture zone and stagnation point were estimated using analytical equations. Once a well location and pumping rate were selected, a simulation with a very short pumping period (.001 yr) was made in order to observe the steady state drawdown. The pumping rate was then adjusted accordingly to meet the drawdown criteria, and a final computer run was made.

Based upon nine observation wells, the time required for the entire aquifer to attain a concentration at or below 100.0, 10.0, 1.0, 0.1, and $< .05$ units was noted. In the case for a single well, at least two additional computer runs were made with different well locations. The run was selected as final for whichever computer run was able to achieve a concentration of $< .05$ units in the shortest time.

This same procedure was then performed for each of the well patterns. Next, the hydraulic conductivity was changed to $10^{-3}$ ft/sec and again each of the well
patterns was tested. In the run made with a hydraulic conductivity of $10^{-3}$ ft/sec, it was observed that the cleanup time decreased exactly by a factor of 10 compared to the runs made with a hydraulic conductivity of $10^{-4}$ ft/sec, indicating a linear relationship between the cleanup time and the pumping rate or drawdown. It was observed that by varying transmissivity and maintaining drawdown, hydraulic gradient and dispersivity constant, the cleanup time was inversely related to the pumping rate (See Section 8.11 Linearity Effects). From this point on, a hydraulic conductivity of $10^{-4}$ ft/sec was used in the simulations. Based upon these findings, it was determined that drawdown, hydraulic gradient and dispersivity were the parameters of interest and that making additional computer runs for a range of hydraulic conductivities was not necessary. Eight possible combinations of these three parameters exist as shown in Table 7.1 and, they represent the generic range of hydrogeologic conditions or aquifers studied in this project.

A series of runs were next made using a drawdown of $\leq 5$ ft (low value), a hydraulic gradient of 0.008 (high value) and a dispersivity of 10 ft (low value), representing aquifer E (Table 7.1). Using these parameters, all of the well patterns were tested, and cleanup times versus concentration were tabulated. The next series of runs consisted of using a drawdown of $\leq 5$ ft (low value), a hydraulic gradient of 0.008 (high value) and a dispersivity of 30 ft (high value) corresponding to aquifer H. All of the well patterns were simulated under these conditions. Only two or three of the seven well patterns were selected and then tested for the remaining generic hydrogeologic conditions. Therefore, all of the well patterns were optimized only under hydrogeologic conditions A, E and H.
<table>
<thead>
<tr>
<th>Condition</th>
<th>Maximum Drawdown</th>
<th>Hydraulic Gradient</th>
<th>Longitudinal Dispersivity</th>
</tr>
</thead>
<tbody>
<tr>
<td>A</td>
<td>High s</td>
<td>Low i</td>
<td>Low a</td>
</tr>
<tr>
<td>B</td>
<td>Low s</td>
<td>Low i</td>
<td>Low a</td>
</tr>
<tr>
<td>C</td>
<td>Low s</td>
<td>Low i</td>
<td>High a</td>
</tr>
<tr>
<td>D</td>
<td>High s</td>
<td>Low i</td>
<td>High a</td>
</tr>
<tr>
<td>E</td>
<td>Low s</td>
<td>High i</td>
<td>Low a</td>
</tr>
<tr>
<td>F</td>
<td>High s</td>
<td>High i</td>
<td>Low a</td>
</tr>
<tr>
<td>G</td>
<td>High s</td>
<td>High i</td>
<td>High a</td>
</tr>
<tr>
<td>H</td>
<td>Low s</td>
<td>High i</td>
<td>High a</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Conditions</th>
</tr>
</thead>
<tbody>
<tr>
<td>High s = ≤ 10 ft</td>
</tr>
<tr>
<td>Low s = ≤ 5 ft</td>
</tr>
<tr>
<td>High i = .008</td>
</tr>
<tr>
<td>Low i = .0008</td>
</tr>
<tr>
<td>High a = 30 ft</td>
</tr>
<tr>
<td>Low a = 10 ft</td>
</tr>
</tbody>
</table>

**Table 7.1** Different generic hydrogeologic conditions modeled.
8.0 RESULTS

8.1 Well Location

Cleanup time is a function of the velocity field which is governed by both the locations of wells with respect to the contaminant plume and the regional flow field. The resultant contaminant velocity is the sum of the velocity produced by the pumping well plus the natural flow in the aquifer. Under a low natural flow, it would not be practical to site a single extraction well at the downgradient edge of a plume, as it would take a long time for the contaminant to reach the well. Under a high natural flow, however, it might be advantageous to place the extraction well near the downgradient edge of a plume. If the well is placed near the upgradient side of the plume, contaminant flow toward the well would be countered by the high natural flow away from the well and contaminant may become trapped in a zone of low flow, or a stagnation area.

Significant differences in cleanup time were observed using different well locations for a given well pattern. Table 8.1 illustrates the impact that a well location has on cleanup time for a single extraction well with a low hydraulic gradient. A fifty foot difference in location added up to a year in the overall cleanup time. In this case, run 1a was selected as final because it achieved a concentration of < .05 units in the shortest time.
<table>
<thead>
<tr>
<th>WELL LOCATION</th>
<th>CLEANUP TIME</th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>col, row</td>
<td>C/C₀</td>
<td>time (yr)</td>
</tr>
<tr>
<td></td>
<td>8 12</td>
<td>1.0</td>
<td>0.0</td>
</tr>
<tr>
<td></td>
<td></td>
<td>0.1</td>
<td>0.54</td>
</tr>
<tr>
<td></td>
<td></td>
<td>0.01</td>
<td>1.45</td>
</tr>
<tr>
<td></td>
<td></td>
<td>0.001</td>
<td>2.52</td>
</tr>
<tr>
<td></td>
<td></td>
<td>0.0001</td>
<td>3.36</td>
</tr>
<tr>
<td></td>
<td></td>
<td>&lt;0.00005</td>
<td>3.96</td>
</tr>
<tr>
<td></td>
<td>8 13</td>
<td>1.0</td>
<td>0.0</td>
</tr>
<tr>
<td></td>
<td></td>
<td>0.1</td>
<td>0.60</td>
</tr>
<tr>
<td></td>
<td></td>
<td>0.01</td>
<td>1.32</td>
</tr>
<tr>
<td></td>
<td></td>
<td>0.001</td>
<td>2.04</td>
</tr>
<tr>
<td></td>
<td></td>
<td>0.0001</td>
<td>2.78</td>
</tr>
<tr>
<td></td>
<td></td>
<td>&lt;0.00005</td>
<td>3.19</td>
</tr>
<tr>
<td></td>
<td>8 14</td>
<td>1.0</td>
<td>0.0</td>
</tr>
<tr>
<td></td>
<td></td>
<td>0.1</td>
<td>1.10</td>
</tr>
<tr>
<td></td>
<td></td>
<td>0.01</td>
<td>1.88</td>
</tr>
<tr>
<td></td>
<td></td>
<td>0.001</td>
<td>2.73</td>
</tr>
<tr>
<td></td>
<td></td>
<td>0.0001</td>
<td>3.53</td>
</tr>
<tr>
<td></td>
<td></td>
<td>&lt;0.00005</td>
<td>&gt;4.00</td>
</tr>
</tbody>
</table>

\( Q_w = 0.012 \text{ cfs} \)
\( s = \leq 10 \text{ ft} \)
\( i = 0.0008 \)
Long. dispersivity, \( a_1 = 10 \text{ ft} \)
\( K = 0.0001 \text{ ft/sec} \)
\( b = 10 \text{ ft} \)

Table 8.1 Effect of well location on cleanup time.
8.2 Water Requirements

By selecting the run with the shortest cleanup time, the volume of water requiring treatment is minimized. However, this is not the absolute minimum volume of water necessary to achieve the cleanup. In order to minimize the amount of water requiring treatment it would be necessary to create a perfect capture zone encompassing only the contaminant plume. By creating an ideal capture zone, the cleanup time is increased significantly because of the lower pumping rates required. In this study, the withdrawal schemes focused on attaining a rapid cleanup through a prescribed maximum level of drawdown, rather than the creation of a perfect capture zone.

Equations 6.2 and 6.5 can be used to compute the pumping rates that would be just large enough to capture a plume for a single and doublet, respectively. Figure 8.1 shows two capture zones computed for a single well at two different flow rates. Two important points are illustrated in this Figure 8.1. First, to completely remove the contaminant plume, the extraction well will have to accept some uncontaminated water from upgradient. Using the outer capture zone in Figure 8.1, there are approximately 26 cells containing zero contaminant that would be accepted into the well before the last of the contaminant is withdrawn. This results in a minimum of 1.5 pore volumes being treated. Second, the use of a finer grid spacing is advantageous because the groundwater divide bisects several contaminated grid cells. In designing an ideal capture zone, dispersive mixing across groundwater flow lines and aquifer heterogeneities which are not included in the analytical equations, are problems which should be addressed. In this study, dispersion and aquifer heterogeneities were both easily accommodated for by using a numerical technique, such as the USGS MOC Model.
Figure 8.1 Single well capture zones for different pumping rates based upon Equation 6.1.
Using a simple advection model predicts that the volume of water that must be removed is equal to the volume of contaminated water initially present. Consider an aquifer of unit thickness, b, containing a circular area of contamination. A single pore volume of water must be removed to reduce the concentration in the aquifer to zero. One pore volume is equal to $\pi/4d^2 n$ or $\pi r^2 n$, where r is the radius of the circular zone of contamination and n is the effective porosity. To pump an additional 3 pore volumes or a total of 4 pore volumes using a single extraction well, the maximum radial distance from which the water is drawn into the well is equal to 2r. It can be shown that, under ideal radial flow conditions, the radial distance from which water must be drawn is equivalent to the square root of the total number of pore volumes pumped.
8.3 High s, Low i, Low α - Hydrogeologic Condition A

Under hydrogeologic condition A, all of the cleanup schemes were simulated and results are given in the Appendix. In terms of final cleanup time (time required for the entire aquifer to achieve a relative concentration of <.00005), the pattern with the greatest number of wells, double triangle, takes the least time, but the difference (.07 yr) between the double triangle and double-cell is insignificant (Figure 8.2a). Overall, the double-cell pattern has the best track record for cleanup time. The single well takes the most time to attain the final cleanup criteria. However, a relationship does not exist between the number of wells in a pattern and the length of time for a certain level of cleanup. For example, the five-spot falls in next to last place behind the single well for final cleanup time. In this study, the results for the 5-spot well pattern were very discouraging. The poor performance is attributed to the limited hydraulic benefits of the four injection wells. There is simply not enough water to provide any significant hydraulic effects once water from the pumping well is equally distributed among the four injection wells. The 5-spot pattern could probably be improved if more of the water was distributed to the upgradient wells and/or an additional source of water were available. In the petroleum industry, multiple 5-spot patterns are employed which results in an equal number of production and injection wells.

Except for the 5-spot pattern, the withdrawal schemes having both pumping and injection wells attain final cleanup more quickly than the single and centerline well patterns which utilize only pumping wells. This results from the steeper hydraulic gradient created by the injection wells. The benefits of a decreased cleanup time using injection wells is countered by an increase in the volume of water circulated (Figure 8.3). Overall, there is not a very significant
Figure 8.2b
Relative Concentration vs. Cleanup Time
High Q, Low i, Low Disp.

\[ T = 10^{-4} \text{ ft}^2/\text{sec} \]
Figure 8.3
Relative Concentration vs. Volume of Water Circulated
High Q, Low I, Low Disp.

$T = 10^{-3} \text{ ft}^2/\text{sec}$

- double triangle
- double cell triangle
- single
- centerline

Volume Circulated $(\text{ft}^3 \times 10^5)$

$0$ $5$ $10$ $15$ $20$ $25$ $30$ $35$

$0.0000$ $0.0001$ $0.001$ $0.01$ $0.1$ $1$

$C/Co$
difference in final cleanup time among the different schemes with injection wells at a transmissivity of $10^{-3}$ ft$^2$/sec. However, in lower transmissivity aquifers, the differences in cleanup time are multiplicative (See Section 8.11 Linearity Effects), and pumping costs could become a substantial factor. Relative concentration versus cleanup time for an aquifer having a transmissivity of $10^{-4}$ft$^2$/sec (Figure 8.2b) is shown for comparison.

Determining the best well pattern for a groundwater cleanup is highly site-specific. Such factors as contaminant concentration, treatment process, size of treatment facility, maintenance, pumping costs, sampling and monitoring to name a few, need to be evaluated. Based upon the results obtained in this study, the 3-spot, doublet and double-cell appear to be the most promising well patterns for a high drawdown, low hydraulic gradient and low aquifer dispersivity. These well patterns minimize the total volume of water circulated, volume of water treated (Figure 8.4) and cleanup time. The double-cell is the most efficient well pattern for achieving up to a 99.9% reduction in contaminant concentration. Beyond this requirement, the volume of water circulated using the double-cell increases significantly.
Figure 8.4
Relative Concentration vs. Volume of Water Treated
High Q, Low i, Low Disp.

C/Co

$T = 10^{-3} \text{ ft}^2/\text{sec}$

Volume Treated ($\text{ft}^3 \times 10^5$)
8.4 Low s, Low i, Low α - Hydrogeologic Condition B

Under hydrogeologic condition B (Table 7.1), the single, doublet and double triangle patterns were simulated using the same well locations that were optimized in condition A. The only difference between condition A and condition B is the smaller drawdown under condition B. In order to achieve a drawdown of \( \leq 5 \) ft, the pumping rates used in condition A were decreased by 50%. Cleanup times were observed to increase almost exactly twice that of the cleanup times in condition A. This indicates that contaminant transport is dominated by advective flow under both high and low drawdowns. Based upon this finding, Figure 8.2, showing relative concentration versus cleanup time for condition B, is applicable to condition A if the time (abscissa) is multiplied by a factor of two. Figures 8.3 and 8.4, showing relative concentration versus volume of water pumped and circulated, are applicable as is because the increase in cleanup time is exactly cancelled out by the decrease in pumping rate so that the volume of water is conserved.

8.5 Low s, Low i, High α - Hydrogeologic Condition C

The double triangle pattern was tested under hydrogeologic condition C. Final cleanup time increased four-fold compared to condition A due to the lower drawdown used with the higher aquifer dispersivity for condition C.
8.6 High s, Low i, High α - Hydrogeologic Condition D

Using the same well locations as condition A, the doublet and double triangle patterns were simulated under hydrogeologic condition D. As a result of the large amount of initial dispersive mixing in condition D (α = 30 ft), less time was required to achieve a 90% reduction in contaminant concentration than with the lower dispersivity (α = 10 ft) used in condition A. However, increasingly longer times than required for condition A, were needed to achieve any further reductions in concentration as shown in Table 8.2. Under hydrogeologic condition D, attaining a 99.995% reduction in concentration takes twice as long as condition A.

8.7 Low s, High i, Low α - Hydrogeologic Condition E

Under hydrogeologic condition E, all of the cleanup schemes were simulated and results are given in the Appendix. The effect of a high hydraulic gradient (.008) is illustrated in Table 8.3, which compares final cleanup time and the number of pore volumes treated for condition E and condition B (low s, low i, low α). With a higher hydraulic gradient, more time and additional pore volumes are required to achieve final cleanup than with the lower hydraulic gradient. The double triangle well pattern is an exception. The hydraulic gradient has no effect upon the double triangle pattern because the flow regime is so strongly dominated by the well field.
### EFFECT OF DISPERSIVITY ON CLEANUP TIME

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<td>.0001 &lt; .00005</td>
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Hydrogeologic Condition A - High s, Low i, Low α 
Hydrogeologic Condition D - High s, Low i, High α

**Table 8.2** Effect of dispersivity on cleanup time.
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<td>4.16 (4.4)</td>
<td>4.15 (4.3)</td>
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</tbody>
</table>

P.V. = Number of Pore Volumes Requiring Treatment
Hydrogeologic Condition B - Low s, Low i Low α
Hydrogeologic Condition E - Low s, High i, Low α

Table 8.3 Effect of hydraulic gradient on cleanup schemes.
Under both a low and a high hydraulic gradient, the double triangle pattern takes the least amount of time to achieve final cleanup (Figures 8.2 and 8.5); however, the total volume of water circulated is larger than any other well pattern studied (Figure 8.6). The doublet which performs very well under low hydraulic gradient conditions is ineffective under a high hydraulic gradient.

In practice, the volume of water pumped using the centerline pattern could actually be less than the amounts shown in Figures 8.4 and 8.7. Throughout the entire simulation, all three wells were left turned on, even after the contaminant plume had passed downgradient of the upgradient injection well. This upgradient well would in all likelihood be turned off. A better plan would be to convert it into an upgradient injection well once the plume has passed by.

The 3-spot is the preferred well pattern under hydrogeologic condition E. Constrained by the maximum allowable drawdown, containing the contaminant plume under this high hydraulic gradient is difficult. The downgradient injection well in the 3-spot, however, provides for a distinct advantage over the other well patterns. The 3-spot pattern requires the least volume of water for treatment (Figure 8.7) and is second to the double triangle in cleanup time (Figure 8.5). And the volume of water circulated is the least among the well patterns involving pumping and injection.
Figure 8.5  
Relative Concentration vs. Cleanup Time  
Low Q, High i, Low Disp.  

\( T = 10^{-3} \text{ ft}^2/\text{sec} \)
Figure 8.6
Relative Concentration vs. Volume of Water Circulated
Low Q, High i, Low Disp.

\[ T = 10^{-3} \text{ ft}^2/\text{sec} \]

Graph showing the relationship between Relative Concentration (\(C/C_0\)) and Volume Circulated (\(\text{ft}^3 \times 10^5\)). Different lines represent single, double triangle, 3-spot, 5-spot, and double cell configurations.
Figure 8.7
Relative Concentration vs. Volume of Water Treated
Low Q, High i, Low Disp.
8.8 High s, High i, Low α - Hydrogeologic Condition F

Under hydrogeologic condition F, the centerline, 3-spot and doublet well patterns were simulated using the well locations optimized under condition E. Pumping rates were increased 100% in order to attain twice the drawdown of condition E. With the higher pumping rates, cleanup times decreased by approximately 60%. It is interesting to note, that with a low hydraulic gradient, doubling the pumping rates decreased cleanup times by 50%. This result suggests that the hydrodynamic dispersion mechanism of contaminant transport is significant at higher hydraulic gradients. Figure 8.5 shows relative concentration versus cleanup time for condition E and is applicable under condition F if the time (abscissa) is multiplied by a factor of 0.6. Figures 8.6 and 8.7, showing relative concentration versus volume of water pumped and circulated, are both applicable to hydrogeologic condition F if the volume of water (abscissa) is multiplied by 1.2.

8.9 High s, High i, High α - Hydrogeologic Condition G

The single, doublet and 3-spot well patterns were simulated under hydrogeologic condition G. Because of the higher dispersivity, a change in the well locations from those used in condition E for both the single and doublet patterns was necessary to contain the plume. To achieve 90% reductions in contaminant concentrations with both the single well and the doublet took approximately 40% of the times required under condition E. To achieve 99.995% reductions (final cleanup) took 90% of the times in condition E. The well locations for the 3-spot were the same as those used in condition E, and achieving 99.99% and 99.995% reductions required the same amount of time as the same pattern in condition E,
which is still less than the time required for either the single or doublet patterns.

8.10 Low s, High i, High α - Hydrogeologic Condition H

All the well patterns, except for the double-cell and 5-spot, were simulated using various well locations under hydrogeologic condition H. None of the withdrawal schemes were successful in containing the pollutant plume. With each well pattern, high concentrations of contaminant was observed in observation well (8,20) located fifty feet from the downgradient edge of the initial plume. Under low drawdown, high hydraulic gradient and high dispersivity conditions, none of the well patterns were able to produce a large enough capture zone to contain the plume.

8.11 Linearity Effects

For a given withdrawal scheme, by varying transmissivity and maintaining one of the sets of hydrogeologic conditions (Table 7.1) as constant, the cleanup time was found to be inversely related to the pumping rate. For example, to achieve a dimensionless concentration of < 100.0 in an aquifer for a given hydrogeologic condition (Base Run) takes one year. Maintaining the same cone of depression, hydraulic gradient, and aquifer dispersivity as the Base Run and decreasing the transmissivity ten-fold, the required pumping rate will be decreased ten-fold according to the Thiem equation (Eq 5.3). It was observed that the cleanup time increases exactly ten-fold from one year to ten years. Also, the areal distribution of the contaminant plume after ten years is exactly identical to the contaminant plume with the higher transmissivity at one year (Base Run). Dispersion has minimal effects. Figure 8.8 illustrates the linearity effect on a
Figure 8.8 Linearity effect on a plume cleanup.
plume cleanup. This finding can be summarized by the following observation: Cleanup time is inversely related to the pumping rate if the Q/T ratio or cone of depression is kept constant. This finding has been verified by the Wilson and Miller (1978) 2-D analytical solution of the advection-dispersion equation.

Several arguments could be made to validate the linearity finding, however, probably the least compelling argument is that it has been observed. The mathematical underpinnings of the linearity lies with the advection-dispersion equation. In the absence of geochemical reactions, variations in concentration over time are the result of the processes of advection and dispersion. By varying transmissivity, the advection term changes, however, the flow field remains unchanged. Thus the solute particles are advected along the same identical flow paths but are displaced in time corresponding to the change in transmissivity. Transport due to dispersion is affected in a similar way. For the example shown in Figure 8.8, the amount of dispersion, after one year, is reduced from that of the Base Run, corresponding to the ten-fold decrease in velocity. After a period of ten years, the amount of dispersion in the lower transmissivity run is equal to that of the Base Run.

Figure 8.9 illustrates an application of the linearity to a 3-spot withdrawal scheme. The hydrogeologic parameters include a high drawdown, low hydraulic gradient and low dispersivity. Each concentration versus cleanup time curve corresponds to an aquifer having a transmissivity differing by a factor of ten. To maintain the same cone of depression in each aquifer, the pumping rates differ by a factor of ten according to the Thiem equation (Eq 5.3). Note how the time required to attain the same desired level of cleanup in each aquifer is displaced exactly by a factor of ten and is inversely proportional to the pumping rate required to maintain the same cone of depression in each aquifer.
Figure 8.9 Application of linearity to a 3-spot well pattern

High Q, Low i, Low Disp.

\[ T = 10^2 \text{E-2} \quad \text{(ft}^2/\text{sec)} \]

\[ T = 10^3 \text{E-3} \quad \text{(ft}^2/\text{sec)} \]

\[ T = 10^4 \text{E-4} \quad \text{(ft}^2/\text{sec)} \]

\[ T = 10^5 \text{E-5} \quad \text{(ft}^2/\text{sec)} \]

\[ C/Co \]

\[ \text{Time (years)} \]
The number of simulations was reduced dramatically as a result of the linearity effect. A single value for transmissivity \(10^{-3} \text{ ft}^2/\text{sec}\) was used in the simulations instead of making a computer run for each cleanup scheme for each value of transmissivity. Results for a withdrawal scheme under a given set of hydrogeologic conditions (Table 7.1) can be extrapolated to an aquifer having any value of transmissivity. For example, Figures 8.2 and 8.5 showing relative concentration versus time, can be extrapolated to any value of transmissivity by multiplying the abscissa by \(10^3(\text{ft}^2/\text{sec})/\text{transmissivity}\). This manipulation requires that the same hydrogeologic conditions -- drawdown, hydraulic gradient and aquifer dispersivity -- be maintained. Similarly, Figures 8.2 and 8.5 can be extrapolated to any transmissivity under hydrogeologic conditions B and F, if the abscissa is first multiplied by a factor of 2 or 0.6, respectively. Figures 8.3 and 8.6, showing relative concentration versus volume of water circulated, and Figures 8.4 and 8.7 showing relative concentration versus volume of water pumped, are applicable to an aquifer with any value of transmissivity and require no manipulation. This is readily apparent if one considers that in order to maintain a constant cone of depression, any change in transmissivity is offset by a corresponding change in the pumping rate according to the Thiem equation (Eq 5.3). Figures 8.3 and 8.4 are also applicable as is for application to hydrogeologic condition B (See Section 8.4 Hydrogeologic Condition B). For application to hydrogeologic condition F, the abscissa in both Figures 8.6 and 8.7 needs to be multiplied by 1.2.
For solutes undergoing linear reversible equilibrium sorption-desorption, the retardation factor $R_p$, is defined as the velocity of solute divided by the average linear velocity of the ground water. This retardation factor can be used to estimate the cleanup time for an adsorbed species by multiplying the abscissa by $R_p$ in Figures 8.2a and 8.5.
9.0 FIELD SITE APPLICATION

To evaluate the results obtained from the hypothetical modeling study, an actual field site was selected for designing a withdrawal scheme. A recovery well system is already in operation at this site. A detailed hydrogeologic description along with a discussion on the contaminant plume delineation study using the USGS MOC Model is given in Freeberg et al., (1987).

It has been estimated that 8 kg of trichloroethylene (TCE) has leaked from an underground storage tank into an underlying shallow sandy aquifer. The underground storage tank has subsequently been excavated, however, a source of contamination still persists in the unsaturated zone along with a number of other point sources within the industrial complex. The contaminant plume is nearly circular and encompasses an area of 11.5 acres (500,000 ft²). The maximum TCE concentration measured in the plume prior to the start up of the recovery system in April, 1984 was approximately 1022 µg/l. In September, 1986, the latest round of sampling for which data is available, the maximum TCE concentration measured was 50 µg/l.

The recovery well system in operation at the site today, consists of 4 pumping wells (previously used as monitoring wells) located along the axis of the plume in the direction of flow. Wells average between 0.011-0.033 cfs (5-15 gpm). Records indicate that at any one time, a total not exceeding 0.066 cfs (30 gpm) is pumped from the aquifer. Drawdowns on the order of 1 to 2 feet are observed.

The constraints used for the testing of the different well patterns for this site differed from the constraints that were used previously in the generic modeling study. In order to simulate actual site conditions, a maximum pumping rate of 0.022 cfs (10 gpm) for a single well and a total pumping rate of 0.066 cfs (30
gpm) was set. This is in contrast to the generic modeling study, where the pumping rates were dictated by the maximum prescribed level of drawdown. In addition, other significant differences exist between the field site and the generic sites modeled. Most notable are the contaminant plume size and shape and the heterogeneous flow field. The size of the model grid also differed. The field site was simulated using a 20 x 35 cell grid (15 x 20 cell grid used in generic study) with each cell representing a 50 x 50 ft area, as described in Freeberg et al., (1987). This leads to a significantly larger number of possible well locations to be evaluated:

Using the same previously calibrated hydrogeologic and source loading parameters as Freeberg et al., (1987), the initial contaminant plume was created and subsequently used at the start of each cleanup simulation. The simulations were ended when the maximum contaminant concentration in the aquifer did not exceed 1.0 µg/l, corresponding to a 99.9% reduction in concentration.

In over 20 computer runs, five modified well patterns were tested including the centerline, doublet, 3-spot, double-cell and double triangle. Several of the well patterns were modified so that each well pattern had a cumulative pumping rate of 0.066 cfs (30 gpm) and no single well exceeded 0.022 cfs (10 gpm). The same conditions were applied to injection wells. The well patterns and associated pumping rates are shown in Figure 9.1.
Figure 9.1 Well patterns and associated pumping rates.
The centerline was the first well pattern investigated. The wells were initially placed equidistant along the centerline of the plume. Then, based upon the contaminant map output from the model, well locations were changed in order to cleanup those areas in which the contaminant concentrations appeared to remain high and were not being effectively reduced. This same general procedure was used for each of the well patterns. Because of the heterogeneous flow field and the large number of possible well locations, it was very difficult to optimize the well locations for any given well pattern.

Table 9.1 shows the length of time required for each well pattern to reduce the maximum contaminant concentration in the aquifer to $\leq 1.0 \ \mu g/l$. The best doublet well pattern was very similar to the double triangle and only one well differed in location by 100 feet. Both the 3-spot and double-cell well patterns are very ineffective in cleaning up the plume as seen in Table 9.1. The poor performance of these well patterns is attributed to the manner in which the 0.066 cfs (30 gpm) is distributed. Both these patterns have a large number of wells, and also contain wells with pumping rates of 0.011 cfs (5 gpm), which have limited hydraulic benefits. The double triangle and centerline patterns performed the best. Both of these patterns did not have to be modified for this study, and are the same identical patterns used in the generic study. It is interesting to note, that the centerline pattern was adopted by the engineering firm hired to design a recovery well system at the site.
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Table 9.1 Time required to achieve a concentration of 1.0 μg/l.
10.0 CONCLUSIONS AND RECOMMENDATIONS

The USGS MOC Model is a useful tool for evaluating different well patterns in an aquifer restoration scheme under variable hydrogeologic conditions. The best well pattern for a groundwater cleanup is highly site-specific and depends upon the objectives and constraints for each problem. The key hydrogeologic variables which control the rate of cleanup are well locations, pumping rates, transmissivity, dispersivity and hydraulic gradient. For a given set of well locations, by varying transmissivity and maintaining drawdown, dispersivity and hydraulic gradient constant, the cleanup time was found to be inversely related to the pumping rate.

Seven well patterns were evaluated under different common hydrogeologic conditions on the basis of cleanup time, volume of water circulated and volume of water requiring treatment. The following conclusions can be drawn from the analysis:

1) Significant differences in cleanup time were observed using different well locations for a given well pattern. Selecting the well locations with the shortest cleanup time also minimizes the volume of water requiring treatment.

2) The 3-spot, doublet and double-cell well patterns are effective under low hydraulic gradient conditions. These well patterns minimize cleanup time, volume of water circulated and volume of water treated. These well patterns require on-site treatment and reinjection.

3) The 3-spot performed better than any of the other well patterns studied under a high hydraulic gradient, high drawdown and either a low or a
high dispersivity.

4) None of the well patterns investigated were able to contain and cleanup the contaminant plume under a high gradient, low drawdown and high dispersivity.

5) The centerline well pattern is effective in achieving up to a 99% level of contaminant reduction under both low and high gradient conditions, but may present a water disposal problem.

6) The 5-spot well pattern performed poorly in this study.

Based on this study and conclusions, it is recommended that the 3-spot well pattern be considered in the design of an aquifer restoration scheme. When dealing with other sites and different plume sizes and shapes and other complexities including cleanup criteria, constraints and heterogeneous flow fields, these results may not be applicable and other well patterns should be investigated. In situations where cleanup time is of paramount importance, bar no expense, a combination of the double triangle and centerline well patterns may be very effective.
11.0 REFERENCES


Freyberg, D. L., 1986, A natural gradient experiment on solute transport in a


Keely, J.F. and C.F. Tsang, 1983, Velocity plots and capture zones of pumping
centers for ground-water investigations, *Ground Water*, 21, pp 701-714.


Wagner, K. and Z. Kosin, 1985, In situ treatment , in *Sixth National Conference*


## WELL PATTERN SUMMARY

High s, Low i, Low disp.

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## WELL PATTERN SUMMARY

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### WELL PATTERN SUMMARY

High s, Low i, Low disp.

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