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UNIVERSITY OF ALBERTA

A CRITICAL EVALUATION OF PUMP-AND-TREAT REMEDIATION TECHNOLOGY AND METHODOLOGY

BY

BRENDAN MARC ADAMS ©

A thesis submitted to the Faculty of Graduate Studies and Research in partial fulfillment of the requirements for the degree of Master of Science.

DEPARTMENT OF CIVIL ENGINEERING

EDMONTON, ALBERTA
SPRING, 1995

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UNIVERSITY OF ALBERTA

FACULTY OF GRADUATE STUDIES AND RESEARCH

The undersigned certify that they have read, and recommend to the Faculty of Graduate Studies and Research for acceptance, a thesis entitled **A Critical Evaluation of Pump-and-Treat Remediation Technology and Methodology** submitted by Brendan Marc Adams in partial fulfillment of the requirements for the degree of **Master of Science**.

Dr. N. R. Morgenstern (Supervisor)

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Dr. C. A. Mendoza

Date: January 26, 1995

ABSTRACT

There are numerous interacting elements within the contaminant transport regime that have critical effects on the behaviour of constituents of concern. Complete and accurate characterization of these components is generally impractical, particularly in heterogeneous regimes. Uncertainty is therefore a reality that must be accepted in the design and operation of pump-and-treat remediation systems.

Empirical knowledge, with respect to full scale application of pump-and-treat remediation systems, is best summarized by a review of case histories. The published case histories available for review do not reveal a discernible trend of system response versus geology, hydrogeology, or contaminant properties. Heterogeneity of the site and inadequate source removal are cited as two common causes of difficulty in achieving remediation success.

Linking scientific knowledge with an empirical database follows from this exercise. Inference, prediction and direct determination are supplied as three methods of characterization of various aspects of a site. A modelling case study illustrates the application of inference methods in this respect.

Uncertainty is ubiquitous in subsurface remediation problems. The question of how to manage this reality follows from acceptance of the low probability of complete characterization in the design phase. A methodology is suggested that accepts uncertainty and responds to it in a structured manner; anticipating possible deviation from previous site conceptualizations and accommodating their demands into the project design.

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TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

LIST OF NOMENCLATURE

NAPL	nonaqueous-phase liquid
DNAPL	dense nonaqueous-phase liquid
LNAPL	light nonaqueous-phase liquid
CERCLA	Comprehensive Environmental Response, Compensation, and Liability Act
RCRA	Resource Conservation and Recovery Act
TCE	trichloroethene
MCL	maximum contaminant level
TCA	1,1,1-trichloroethane
DBCP	dibromochloropropane
U. S. EPA	United States Environmental Protection Agency
NEPA	National Environmental Policy Act
CV	coefficient of variation
σ	absolute standard deviation
σ^2	absolute variance
μ	absolute mean
F	specified percentage variation in the sample mean
N	number of samples (dependent)
α	probability of a given variation in the mean
β	probability of detection of a variation in the mean
$Z_{\alpha/2}$	upper /2 fraction of the standard normal distribution
Z_{β}	upper fraction of the standard normal distribution
μ_Y	absolute mean in log-normal distribution
σ_Y^2	absolute variance in log-normal distribution
μ_X	absolute mean in normal distribution
σ_X^2	absolute variance in normal distribution
E[ ]	expectation of given variable
e	base of normal logarithm
V_X^2	coefficient of variation squared
K_o	saturated hydraulic conductivity
θ	volumetric water content
$K(\theta)$	unsaturated hydraulic conductivity at q water content
$\beta(\theta)$	unsaturated hydraulic conductivity parameter at q water content
θ_o	saturated volumetric water content

S_D	standard deviation of sample group
var[ ]	variance of a variable
$\overline{X}$	mean of a sample group
n	sample group size (independent)
S_D^2	variance of a sample group
$\sigma^2_{\overline{X}}$	absolute variance in a sample group mean
$\sigma^2_{S^2}$	absolute variance in a sample group variance
REV	representative elemental volume
I_h	correlation (or integral) scale
I_Y	log-correlation (or integral) scale
D	dimension of averaging volume
L	extent of flow domain
SPT	standard penetration test
CPT	cone penetrometer test
D_{10}	10% finer-than grain size
K	hydraulic conductivity
K_{ef}	effective hydraulic conductivity
K_G	geometric mean hydraulic conductivity
J	head gradient
σ_ϕ^2	variance of effective hydraulic conductivity
CEC	cation exchange capacity
f_{oc}	organic carbon fraction
ASTM	American Society for Testing and Materials
K_{ow}	octanol-water partition coefficient
K_d	distribution coefficient
K_{ij}	hydraulic conductivity tensor
S_s	storativity
R	retardation factor
D_{ij}	hydrodynamic dispersion coefficient tensor
h	hydraulic head
v_i	velocity vector
c	concentration
G	source/sink term
K_{sat}	saturated hydraulic conductivity
λ	decay constant

PCE	tetrachloroethene, perchloroethene
1,1,1-TCA	1,1,1-trichlorothane
K_{oc}	organic carbon partition coefficient
gpd	gallons per day
ft/day	feet per day
K_v	vertical hydraulic conductivity
1,1-DCE	1,1-dichloroethene
cis-1,2-DCE	*cis*-1,2-dichloroethene
trans-1,2-DCE	*trans*-1,2-dichloroethene
VOC	volatile organic compounds
µg/l	micrograms per liter (ppb)
BTEX	benzene, toluene, ethylbenzene, and xylenes
VES	vapour extraction system
1,2-DCA	1,2-dichloroethane
GAC	granular activated carbon
PRP	potentially responsible parties
JP4	military jet fuel mixture
S_y	specific yield
S_G	specific gravity
σ_{IFT}	interfacial liquid tension
S_o	solubility of liquid in water (at 10°C unless otherwise noted)
vp	vapour pressure
H	Henry's constant
K_{oc}	organic carbon partitioning coefficient
ν	kinematic viscosity (at 20°C unless otherwise noted)
MCL	maximum contaminant levels
IGWMC	International Ground Water Modelling Center
MMOC	modified method of characteristics

CHAPTER 1: INTRODUCTION

1.1 INTRODUCTION

The field of contaminated site remediation has been approached from many different directions in the past few decades. These approaches include those from the sciences (chemistry, biology, soil science, hydrogeology, hydrology, environmental sciences, etc.) and engineering (geotechnical, environmental, chemical, etc.).

Morgenstern (1991) described the emergence of a field, termed 'environmental geotechnics', which encompassed the geotechnical aspects of environmental remediation and waste management. Many other engineering and scientific fields should be involved in any discussion of remediation and these areas will therefore be addressed in this thesis. However environmental geotechnics, and its unique association with the traditional world of geotechnical engineering, is the central driving force behind the study described in this report.

The approaches differ largely in their approach to uncertainty. Science, in principle, could be hypothesized as the process of eliminating uncertainty. Conversely, engineering strives to manage an inevitable degree of uncertainty in practical systems. A balance must be struck between attainable knowledge that can reduce uncertainty and a manageable level of uncertainty in remediation projects. For remediation of contaminated soil and aquifers, this balance is usually dictated by the most cost effective solution.

The major components in the process of remediating sites have been the contaminants (chemicals), the environment (subsurface regime), the receptors (water supply wells, surface waters, flora, fauna or ultimately the human consumer), and an effective remedial technique that would alter the system in a positive manner. Theory encompassing all of the components in this system would involve a myriad of variables and processes from the entire range of disciplines listed above. Realistically, these variables can only be partially known by the designer, constructor and operator of a remedial system.

1.2 BACKGROUND

1.2.1 Scope of Problem

Since the beginning of civilization, and particularly since the onset of the industrial revolution (circa 1760), society has largely regarded nature as infinite (French, 1990). This has been particulary true in a perception of its ability to absorb the unwanted products and by-products of human activities. A combination of this attitude with the increased concentration of manufacturing and industrial facilities as part of industrialization has led to the environmental scenario that is commonly encountered today. That is, historic environmental loading of the natural systems (i.e. atmosphere, surficial soil, surface water bodies, ground water regimes) beyond their ability to dilute or incorporate the material into the corresponding ecosystem. This history has led to the existence of sites that have been 'contaminated' over a period extending decades or even centuries into the past.

In dealing with pump-and-treat remediation of aquifers, one particular aspect of the above contamination scenarios applies: inadequate handling practices of hazardous materials, and even purposeful disposal methods designed to allow infiltration into the subsurface, have led to distributions of contaminants in the soil and ground water. Several possible scenarios of subsurface contamination that can impact the ground water quality are as follows: existing dissolved mass in ground water subject to further transport and dilution; contaminating material situated above the water-table that might act as a continued source of dissolved-phase contamination (through leaching or vapour-phase transport); and undissolved material in direct contact with the saturated groundwater. This last aspect might be substances of limited solubility, such as immiscible fluids or precipitated solids, or substances adsorbed to soil particles. Rate limitations and other chemical reactions would cause this material to act as a source of dissolved chemicals over even longer periods of time.

Contaminated sites involving organic compounds, have become the center of remedial efforts, particularly pump-and-treat, during the last decade. This is especially true of those polluted by chlorinated solvents or halogenated aliphatic compounds, thought to be the primary contaminant of concern in the late 80's and 90's as petroleum hydrocarbons were in the 80's (Hasbach, 1993). Unfortunately, remediation projects involving immiscible fluids in the subsurface have bought to the forefront previously unencountered problems, particularly those associated with multi-phase transport and mobile source materials. A typical case study illustrating the difficulties of dense nonaqueous-phase liquid (DNAPL) remediation was presented by Herkert and Aslam (1990), where migration of the subterranean DNAPL source pool was postulated to be part of the cause of limited remedial success.

MacKay and Cherry (1989) estimated that hundreds of sites containing what may be considered "major" organic plumes were being addressed under CERCLA and RCRA legislation, and that probably thousands of similar sites in North America exist that are not addressed under these regulatory regimes. Another study (Giampaoli and Bugo, 1991) claimed that private industry spending toward environmental efforts could be as high as one trillion dollars since 1970. This same source stated that Superfund projects in the U.S. may reach the $100 billion cost level over the next five decades.

The 'contaminating potential' of chemicals within the groundwater depends largely on the definition of contamination in terms of an acceptable concentration of the substance in the regime. When dealing with materials of limited solubility, which is significantly above acceptable levels, this potential is usually large. For example trichloroethene (TCE), a chlorinated solvent and commonly considered a carcinogen, is an immiscible fluid when relatively pure with a solubility of 1100 ppm (at 20C). One kilogram of this substance would therefore require a minimum of 909 liters to completely dissolve. Diluting this plume to the MCL value of 5 ppb, for example, would require 200,000,000 liters. It is thus obvious that significant amounts of this chemical in the subsurface would have the capability of contaminating very large volumes of groundwater.

The above example illustrates maximum possible ranges assuming a homogeneous concentration within the plume. MacKay and Cherry (1989) give several examples of existing plumes with total volumes and total dissolved contaminant masses. An extreme example from this source shows 4.5 billion liters of ground water within a plume containing an estimated total of 80 liters of the chemicals TCE, 1,1,1-trichloroethane (TCA), and dibromochloropropane (DBCP). When it is considered that it is usually necessary to extract many times a plume's pore volume in order to significantly alter the ground water quality toward remediation goals, these volumes become even more daunting.

1.2.2 Pump-and-Treat Remediation

Remediation through ground water extraction and treatment is limited in its application to the above contamination scenarios. Pump-and-treat remediation refers to the process of extracting contaminated ground water from the subsurface, bringing it to the surface and treating it to an acceptable level for reinjection or disposal. Therefore the contaminant must enter the dissolved phase in order to be remediated by the system. Alternatively, the extraction system may be designed to capture free-phase liquid contaminants. This could be done independently or in concert with the ground water extraction activity. In either case, the extraction effort can realistically be expected to remove only part of the available contaminant mass due to the complexity of ground water flow fields, particularly in pumped regimes.

Pump-and-treat remediation technology has been applied to contaminated aquifers more than any other remediation technique (Rivett, 1993; MacKay and Cherry, 1989; Hasbach, 1993), excepting perhaps excavation and disposal. Unfortunately, this procedure has seen very limited success in achieving the cleanup goals set for it by governing regulatory bodies (Travis and Doty, 1990). Containment goals have been met in a large percentage of cases (Haley *et al.*, 1991) but this may also be failing in the case of mobile DNAPL sources. Indeed, if the contaminant adsorbs strongly to the soil material, containment may be the only acheivable goal by pump-and-treat actions (Gailey and Gorelick, 1993).

As with many industries, there appears to be a significant gap between state-of-the-art remediation technology and common usage, or what might be termed state-of-the-industry. A review of research papers too numerous to list here revealed that research is very active in the areas of innovative in-situ treatment, enhanced extraction methods, and treatment of extracted material. In-situ technologies commonly center around chemical, biological or biochemical degradation of the substances of concern (eg. biodegradation, reductive dechlorination, dehalogenation), or chemical immobilization of the contaminant through surfactant treatment of the surrounding ground water. Enhanced extraction methods can involve modification of the ground water chemistry to reduce solubility or mobility restrictions (eg. surfactant or solvent mobilization, electrokinetics). It can also involve extraction through phase change (vapour extraction, air sparging, steam flushing) or physical modification of the regime (hydrofracturing) (Hasbach, 1993).

Though many of the above technologies have been investigated in theory and laboratory settings, only a fraction have been successfully transferred to field conditions, and fewer still have been applied to full scale remediation projects. This reality is slowly changing however. With the U.S. EPA's committment to innovative remediation technologies, as outlined in a directive of mid-1992 stating "conventional and/or innovative technologies be considered for enhanced recovery of residual LNAPL and DNAPL from the subsurface" (Hasbach, 1993), these methods may move in the future to complement pump-and-treat techniques and in some cases replace them. The scope of the problem outlined in the preceding text underlines the need for this progress.

Innovative techniques might also be applied to the design and management of pump-and-treat remediation sites. Hoffman (1993) introduced a methodology referred to as "smart pump-and-treat" which included aspects of the observational method and a technique of dynamic well field management. The following thesis introduces another proposed methodology improvement incorporating the observational method in its examination of the broad realities of the field.

1.2.3 Politics of Contamination and Remediation

Often, investigations and proposals from the scientific or engineering fields of research ignore one very dominant aspect of contaminated site remediation. That is the political environment under which the investigation, design, approval, and operation of any system must work. Not only must regulatory regimes be recognized and adhered to, but less formalized input or communication from the public must be considered. In many cases the most expedient course of action, identified through definition of the physical scenario, can be overridden by social or political considerations. This is largely out of the control of the engineer or scientist investigating or designing a remediation project. However, recognition of this reality during the early stages may aid in producing a solution that is not only physically reasonable but also socially and politically acceptable.

Environmental legislation has become a dominant player in the politics of industrialized nations in recent years. King Edward I of England banned the burning of coals found on the seashore, or "sea coles," in a 1306 proclamation that may have been one of the first acts of "environmental legislation." Six hundred years later the Coal Smoke Abatement Society was founded in London to pressure society and government to improve the city's extremely poor air quality (French, 1990). These few examples are typical of the sporadic attention that environmental issues have commanded in the past. Continuous interest in preservation of the environment did not occur until approximately the late 1960's. Since then governments have experienced cycles of attention that have resulted in much of the legislation existing today, particularly in North America.

Efforts in environmental legislation prior to 1970 largely took the form of remedial actions rather than focusing on the prevention of pollution or degradation. Beginning in the early seventies, or possibly with the *National Environmental Policy Act* (NEPA) enacted in 1969 in the United States, both Canada and the U.S. began attempts to regulate and mitigate the degradation of the environment. This marked the onset of "approval based" legislation in North America, which has been copied around the world to differing degrees (Jeffrey, 1989).

1.3 PROBLEM STATEMENT

Pump-and-treat remediation has been one of the most commonly applied remediation technologies in the past two decades. However, the strategy has not been particularly successful (Travis and Doty, 1990; Bredehoeft, 1992). The operations tended to extract much greater ground water volumes than were initially predicted and, even when initial responses to the system were encouraging, the effectiveness fell off over time and goals were not achieved. The extraction system responses might be characterized in the two zones described by Hockman (1992), where the asymptotic contaminant concentration has usually been significantly above remediation goals.

The volume of research into various aspects of the contaminant transport regime over the past few decades has not resulted in more consistently successful designs for pump-and-treat remediation systems. Few design methodologies allow for the uncertainties inherent in these systems and accommodate them in an efficient manner. A brief overview and conceptualization of the state of knowledge in the contaminant transport regime will be introduced in a flowchart in Chapter 2 (Figure 2.1). This will illustrate the major elements of the field, as well as their interrelationships and interactions. Following this, a review of several case histories of pump-and-treat experience will show typical ranges of field conditions and responses. A linking of the scientific knowledge and empirical database provided by these two discussions will lead to an illustrative example of site characterization for pump-and-treat system design through inference and predictive modelling.

The low probability of complete characterization in the design phase needs to be acknowledged and attention shifted to the problem of how the scenario should be managed. A methodology is required that recognizes uncertainty and responds to it in a structured manner, anticipating possible deviations from previous site conceptualizations and accommodating them into the project design. A conclusion regarding this methodology will then be introduced, followed by recommendations for further study.

1.4 THESIS OUTLINE

Following the introduction, a brief background is given of the scope of ground water contamination problems and the considerations that must be made when addressing them. A problem statement follows this discussion. A conceptualization of the contaminant transport regime will then be presented, using a flowchart to illustrate the various interrelationships and interactions. The regime will then be related to the application of remediation in terms of a discussion of remedial investigation considerations. A discussion follows of modelling techniques that could be used in inferences and predictions of those properties and processes that could not be economically determined through direct characterization.

Full-scale pump-and-treat remediation efforts that have been described in published literature will then be reviewed. Eleven sites and their response to remediation will be described in detail. Discussion of the possible reasons for full success, limited success or failure will then follow. A summary will attempt to discern a tendency for the geology or other identifiable characteristics of the case to be related to its degree of success.

The two perspectives of scientific knowledge and empirical data mentioned above will then be related. Inference, prediction and direct determination will be defined as they relate to site characterization for pump-and-treat remediation. An example application of numerical modelling techniques to inference and prediction will then be undertaken. This example will model one of the case studies of Chapter 3 and attempt to extend its characterization through this process.

The conclusion of this report will discuss the problem statement and how the intervening chapters have addressed it. A review of methodologies that could aid in increasing success in pump-and-treat remediation projects will be given. Finally, suggestions for further study arising from the results of this report will be presented.

CHAPTER 2: COMPLEXITY IN THE CONTAMINANT TRANSPORT REGIME

2.1 INTRODUCTION

The first question to be answered when discussing contaminant transport is, what is a contaminant? For the purposes of this thesis, a contaminant shall be considered to be a substance in the ground or ground water that reduces the utility of the ground water for human, animal, or plant consumption. This does not dictate that the substance be made or placed there by people, it could very well be a natural occurrence. However, the usual source of such contamination is anthropogenic.

Of central concern when considering ground water contamination are three issues: what is the toxicity of the material, what is its mobility, and what is its persistence (Jackson *et al.*, 1985)? Discussion of the toxicity of a material crosses into the realm of risk analysis. Though this is a central issue in ground water contamination and is garnering more attention, it is beyond the scope of this study examining the physical aspects of contamination. The ability of a given contaminant to move through the ground water regime is another important issue. Low mobility materials are of lesser concern since they have a smaller chance of affecting large areas of the saturated soil environment. The persist[illegible]aminant is the final important issue. If the material degrades from its toxic state to a more innocuous compound (this commonly applies to organic compounds and to a lesser degree to inorganic compounds), the long time frame of ground water movement makes contact with a receptor unlikely. Essentially, the scenario of most pressing concern is one in which a toxic substance with high mobility and persistence is released into the environment.

Contaminant transport and withdrawal is influenced by numerous interacting properties and processes within the surface and subsurface environments. Variations of each of the parameters within the regime multiplies the complexity enormously. This chapter will outline the components of contaminant hydrogeology in terms of the three areas of concern; the solid matrix, fluid phase (water and air), and the contaminant (in both free fluid and dissolved phases). The application of pump-and-treat remediation technology and its affect on these components will also be discussed.

2.1.1 Property and Process Interrelationships for Contaminant Transport

The flowchart shown in Figure 2.1 illustrates the relationships between various components of the multiphase regime of contaminant hydrogeology. The hierarchy and arrows indicate the relative position and influence of each component in the natural environment. These components might be divided (as shown) into solid matrix, fluid, and contaminant categories, with some elements bordering on two areas. The following subsection explains the details of the figure.

There are three basic methods of determining each of these components for a specific site: direct determination through investigation, inference through interpretation of lower or same level components, or prediction through interpretation of higher level components. Almost all categories might be evaluated to some degree through direct investigation. Some categories can be evaluated quite accurately while others might only be generally identified with further interpretation needed using other components as input. The supposition might be accomplished based on experience and knowledge of the governing processes, or it might be explored through modelling. Commonly, those components with numerous inputs require modelling to handle the many possible combinations and interactions. Some investigation is always needed in order to initially define components on which to base the inferences or predictions.

Geology/geomorphology is almost always defined through inference, based on input evidence from topography, geotechnical and hydrogeologic properties, and occasionally geochemistry (to define source materials). Topography is easily observed from direct surface investigation. Geotechnical properties are usually obtained from direct investigation, though their spatial distribution might sometimes be predicted from the geology/geomorphology (if this has been defined through other sources). Hydrogeologic properties are also often determined through direct investigation, though they are sometimes inferred from hydrogeology or correlated to geotechnical properties. Geomorphology may be used to increase understanding of the hydrogeologic property distributions. Hydrogeology is usually defined through direct measurement of the ground water regime, in combination with the hydrogeology definition. Occasionally, inverse modelling of solute transport cases might provide further understanding of the hydrostratigraphy and hydrogeology.

Climate is easily accessible to direct investigation since it occurs above the ground surface and is relatively independent of the other properties and processes. Local recharge and discharge might be predicted from topography and climate or inferred from the hydrostratigraphy and hydrogeology. Regional hydrogeology might be directly measured through a broad investigation or it may be inferred through the climate and geology/geomorphology. It might also be inferred with respect to the site in question through modelling of the hydrogeology. In terms of pump-and-treat remediation, water is the major fluid for this area. However, contributions of the vapour phase in contaminant migration, delineation, and remediation may also be very important.

The contaminant release can be known through historical data or it might be inferred from aqueous, vapour, immiscible, and adsorbed phase quantification. The chemistry of the pore water is usually measured directly in contamination investigations. It might also be inferred from understanding of geochemistry or inverse modelling of the solute migration. The geochemistry is either directly measured or inferred through solute transport modelling. Solute transport can be predicted through input of hydrostratigraphy, hydrogeology, chemistry and geochemistry as well as information on NAPL sources or in might be inferred through input from historical transport data. The NAPL transport properties are often predicted through geochemistry and hydrogeologic property input, or inferred from solute transport patterns.

Physical attenuation through dispersion (dilution) and diffusion usually contributes to the hydrostratigraphic influence on solute transport. Chemical retardation and attenuation through adsorption and decay (biological and chemical) affects the influences of chemistry, geochemistry and NAPL sources on solute transport. These chemical properties might also affect the NAPL transport through geochemistry. Rate limitations might come into play in diffusion, adsorption and decay processes.

2.1.2 Variability of Parameters

A primary requirement in remediation system design is that the site is characterized to the highest practical level. Characterization, in the remediation sense, includes property and process definition of all components comprising the three areas presented in Figure 2.1. Each of these components can be highly variable, both on their own and through interactions with other elements of the environment. The following subsection attempts to compare the relative variability of individual components. The further complexity of component interactions will be discussed after a more detailed examination of each of these elements has been accomplished.

Site conceptualization for environmental remediation is often of intimidating magnitude due to this large degree of variability in hydrologic and contaminant transport properties. Three approaches to this problem exist, as paraphrased from Gelhar (1993):

- complete delineation of the property distributions within a site

- assumption of equivalent homogeneous property values to be applied throughout a site
- assumption of small scale variability of properties that are random, and application statistical theory to infer their distribution

The first of these options would require unacceptable resources to apply in even mildly heterogeneous regimes. The second approach has been widely applied in the past but contains significant methodological errors which render the approach unacceptable for continued application to environmental contamination problems. The third option remains as the alternative with the most promise in present and future site remediation problems. A fourth option might be to accept the inherent uncertainty of the problem and accommodate it through an iterative design process and responsive system management.

The third method, that of characterizing the variability of geologic and hydrogeologic properties through the assumption of randomness in property distributions, has become quite popular in recent years. Stochastic methods have been widely researched with respect to hydrogeology, most notably by Lynn Gelhar and Gedeon Dagan, and subsequently applied to contaminant hydrogeology. Gelhar (1993) provided a very recent and extensive examination of the field beyond the limited scope presented in this paper. Stochastics have also been applied beyond ground water flow theory to the distribution of other common soil properties. A study by Jury (1986) into the variability of soil properties produced the results presented in Table 2.1 (adapted from Warrick and Yeh, 1989; and Jury, 1986). A more detailed listing of the variability of various soil and hydrogeologic properties is presented in Tables A.1, A.2 and A.3, Appendix A.

The coefficient of variability used in Table 2.1 was defined by Warrick and Neilsen (1980) as follows:

$$CV = \left(\frac{\sigma}{\mu}\right) * 100\% \qquad [2.1]$$

The number of samples required to have a 95% probability of detecting a change of F% in the value of the mean (shown for 10%, 50% and 100% in Table 2.1) can be calculated using the following formula from Jury (1986):

$$N = \left\{ \left(Z_{\alpha/2} + Z_{\beta}\right)^2 \frac{CV^2}{F^2} \right\} + 0.5 * \left(Z_{\alpha/2}\right)^2 \qquad [2.2]$$

Conversion of statistical parameters based on log-normal distribution (μ_Y And σ_Y^2) to parameters based on normal distribution (μ_X And σ_X^2) might be required for properties with wide variations that are usually characterized in log-normal terms, such as hydraulic conductivity. Gelhar (1993) gave the following relationships for this:

$$\mu_X = E[X] = e^{\mu_Y + \sigma_Y^2/2} \qquad [2.3]$$

$$\sigma_X^2 = E\left[X^2\right] - \{E[X]\}^2 = e^{2\mu_Y + \sigma_Y^2} * \left(e^{\sigma_Y^2} - 1\right) \qquad [2.4]$$

$$\frac{\sigma_X^2}{\mu_X^2} = V_X^2 = e^{\sigma_Y^2} - 1 \qquad [2.5]$$

Though Table 2.1 data encompasses a limited number of cases, some trends can be identified. Solid matrix properties (#1 to #4) had relatively small variation, while those relating to dynamic processes (#6 to #11) varied to a much higher degree (Warrick and Yeh, 1989). Equation [2.2] shows that the number of samples to obtain a given confidence interval for the mean was positively related to the square of the coefficient of variation. This was illustrated by the F values supplied in the preceding table.

It should be remembered that sample group statistical parameters ($\overline{X}$, S_D^2) obtained through investigation are only approximations of the population parameters (μ and σ^2) for a site. Gelhar (1993) presented the following standard relationships for variance of both the mean and the standard deviation for sample groups of limited size:

$$\text{var}\left[\overline{X}\right] = E\left[\left(\overline{X} - \mu\right)^2\right] = \frac{\sigma^2}{n} = \sigma_{\overline{X}}^2 \qquad [2.6]$$

$$\text{var}\left[S_D^2\right] = E\left[\left(S_D^2 - \sigma^2\right)^2\right] = \frac{2\sigma^4}{(n-1)} = \sigma_{S_D^2}^2 \qquad [2.7]$$

The above equations illustrate that an increasing number of samples will give an increasingly better approximation of the actual statistical parameters in the field. Basic statistical theory can subsequently define confidence limits from these values, under certain conditions.

2.1.3 Effect of Sample Size on Properties

The concept of a representative elemental volume (REV) has often been used to describe the minimum volume above which the mean value of a property will become representative of the soil. Bear (1972) illustrated this principal of REV size for porosity in his popular figure showing the decreasing oscillations of porosity about the mean as sample size increased, with the 'representative elemental volume' defined as the point at which a steady-state value is reached. Dagan (1986) gave three fundamental scales over which averaging could occur in stochastic characterizations (see Table 2.2). Site contamination investigations could be seen as being undertaken on the local scale, for most parameters of concern. In the past, these investigations have been dependent on laboratory-scale measurements of properties, extended to local or even regional scales in the characterization of a site. In-situ measurements of properties, such as long-term pumping tests to determine average hydraulic conductivity or transmissivity, can extend the determinations closer to the local scale, which is more compatible with the project size. If the REV concept can be extended to apply at each scale, the importance of measuring as close to the scale of concern as possible is self-evident.

The concept of correlation length in variation of site parameters has been discussed in various stochastic characterization studies (Gelhar and Axeness, 1983; Dagan, 1986). It might be simply explained as the linear distance of influence between correlated statistical environments with respect to a given soil property. In stochastic analysis, the (random) distribution of the value a material property in question is biased by the value of the property in surrounding material, as a function of this dimension.

The implication of correlation lengths to site characterization is that representative macroscopic values for a given property might be obtained, by sampling at a spacing smaller than the correlation length (to delineate the local heterogeneity), and averaged over a soil volume. Stochastic theory suggests that a number of discrete samples can be used to characterize the entire volume through statistical parameters with an assumption of constant variability or 'stationarity'. Deterministic theory assumes the variation of these parameters between discrete sample points follows some mathematical relationship (often linear).

Ex-situ samples give values that are only representative of the sample volume itself. Given the present state of sampling technology, this can usually be assumed to be a point value in terms of deterministic or stochastic analyses of site conditions. Disturbance effects commonly recognized in sampling practice should be considered when interpreting results of measurements performed on these samples. Varying quality in soil samples for determination of certain properties has been well recognized in geotechnical practice (Idel *et al.*, 1989). In-situ testing can avoid much of the disturbance associated with sampling, and for some tests can sample volumes far greater than practical ex-situ sample sizes. These sampled volumes may perhaps extend beyond many correlation lengths of the property's variability and give a representative averaged value. Theoretically, the more heterogeneities a test volume encompasses, the lower the variability observed provided that the size of sampling stays within one scale size. Subsequently the lower the variability, the lower the number of required samples to obtain an acceptable confidence interval. This can be thought of in terms of increasing sampled volume being equivalent to increasing sample numbers (n) in equations [2.6] and [2.7] previously.

2.2 SOLID MATRIX PROPERTIES

Solid matrix properties are very important to the processes of ground water contamination and remediation. The overall nature of the subsurface material derives from its origin and depositional environment. This is often termed the geology or geomorphology of the soil profile. The physical make-up of the soil profile proceeding from this has influences on many of the subsequent elements of contaminant hydrogeology as illustrated in Figure 2.1.

The characterization of a site's solid matrix properties has been a common step in past remedial system design efforts. Much of the methodology for these designs has been derived from geological and geotechnical investigation practices; hydrogeological practices for well field design, and site dewatering techniques for geotechnical works. Though past pump-and-treat design methodologies have been viewed as inadequate in recent years (Travis and Doty, 1990; Cherry *et al.*, 1993), the data requirements involved in physical site characterization is now supplemented by other data demands. The following section is an overview of present day understanding of solid matrix properties from the perspective of contaminant transport and withdrawal.

Physical aspects that affect the behaviour of contaminant migration away from a site or toward an extraction system are: geology and geomorphology, topography, geotechnical properties, and hydrogeologic properties. Though geotechnical properties might not directly relate to the processes involved in remediation of a site, their use as correlations to other properties have proven useful at times.

2.2.1 Geology & Geomorphology

The investigation of geology of a site where subsurface contamination is being assessed, and remedial options are being considered, is commonly concerned with the various elements of stratigraphy. The stratigraphic units of a site are usually defined by depositional origin but often parallel variations in geotechnical, geochemical and hydrogeologic properties. This origin of the stratigraphic units is usually conceptualized in terms of its geomorphology or 'study of the form of the earth'. Geomorphology is an interpretation of the physical processes a site has undergone in its geologic past, relating to deposition, weathering, etc. In Canada, and a large portion of the northern hemisphere, many of these processes date from the Quaternary era and glacial action during that time period.

The glacial successions common in the geologic history of the northern hemisphere have led to a characteristic of great variability over short distances of such properties as: grain size, fabric, and strength (geotechnical properties); thickness and geometry (stratigraphy); surface form (topography); and permeability (hydrogeologic properties) (Fookes and Vaughan 1986). Given this quality of abrupt change, the pattern and degree of variation becomes significant, particularly in relation to the migration and withdrawal patterns of contaminants. Understanding the geomorphology of a site can lead to better expectations of the patterns of stratigraphy. This, in turn, can aid in directing the intensity of initial investigation and the development of further investigation plans.

The elements that contribute to the stratigraphy found at a site today are myriad. They include the preglacial geology, climate, glacial action, nature of source materials, glacial retreat or wasting patterns, and post-glacial weathering. Topography, geology (in terms of hydrologic, geotechnical and geochemical horizons) and previous geomorphologic definitions of the surrounding area are usually necessary to accurately interpret this aspect at the site scale. These clues of the formative processes applied to a site can lead to a geomorphologic definition, which in turn can lead to more informed expectations of the subsequent elements.

The sedimentary environments that are common for glaciated areas of Canada can usually be directly related to the last glaciation or to subsequent depositional and weathering periods. These depositional environments include: fluvial and glaciofluvial sand and gravel deposits, glacial tills (lodgment, meltout, and flowed), and lake and marine basin deposits (ice proximal and ice distal) (Fookes and Vaughan 1986). Elements from these environments vary in make-up due to source material and proximity to their origins. In some cases, the profile is dominated by post-glacial successions such as river valley deposits; in other cases, the sequence is almost entirely due to glacial processes of deposition, such as in glaciolacusterine basins. Of course, pre-Quaternary geology can also be found not only in the subsurface but on the ground surface in some areas. It is apparent from this description that when the geomorphology is inferred for a soil profile, the sequence of events could vary over a wide range of depositional environments.

The physical resolution of soil stratigraphy from investigation is dependent on many factors. These include the boring and sampling methods used, the experience and training of the drilling rig operators, and the depositional environment (with the subsequent layer interface resolution). The necessary spacing of boreholes for accurate delineation is dependent on the lateral variability of the stratigraphic elements within a site.

It has been commonly accepted that the near infinite variation of possible combinations of source material, weathering (type and energy), and sequencing of each of these factors, gives no opportunity for generalization of physical distribution of properties according to depositional environment. New understandings of depositional processes, from present-day river delta studies, may make this prediction possible to some degree (Fogg, 1989), though there have been few empirical investigations to date.

2.2.2 Topography

Topography is the surface relief of a site. It has properties of magnitude of relief, rate of elevation change, and pattern or geometry. These aspects of the topography might be used as input to the interpretation of the near surface geology and geomorphology. In this respect the geometry can provide clues as to the long past pre-glacial, glacial, and post-glacial weathering action a site has been subjected to. This in turn might lead to supposition of the types and distributions of material likely to be found upon further investigation. The permeability of the near surface stratigraphic elements, the slope of the surface, the geometry of drainage channels and the precipitation characteristics all affect the recharge patterns into the subsurface.

At the site scale, topography usually has limited variations in pattern. Topography can be initially defined through use of historical data such as air photographs, geological and geographical maps and site plan drawings. Newly acquired information through site inspection and surveys might also be applied.

2.2.3 Geotechnical Properties

Geotechnical properties relating to contaminant transport such as bulk density of the soil, porosity, water content and grain size distribution can be easily defined. Though some of these properties do not directly affect the transport of contamination, it is possible to form a correlation with many of the properties that do. Bulk density is often related to the geochemical properties of the soil which are dependent on soil matter or surface area density within a particular volume (i.e. retardation coefficients). Porosity, water content and grain size distribution are all commonly related to the moisture retention properties of the soil in the vadose zone, as well as to the prediction of hydraulic conductivity. These same properties have also been related to residual NAPL distribution and relative permeability. Porosity and water content also have a direct influence on the hydrogeologic properties of the soil.

Borehole samples and geophysical methods exist for determination of geotechnical parameters both in-situ and ex-situ. Their distribution within a site can sometimes be qualitatively predicted through understanding of the geomorphology. It has been mentioned previously that often the stratigraphic elements parallel significant changes in these geotechnical properties. Thus, the degree and density of variation is usually dependent on the geologic heterogeneity. The lower variability of these properties, and the subsequent lower number of samples required for site definition, has been discussed previously with respect to Table 2.1. Table A.3 also gives some value distributions for these properties, taken from available literature.

Geotechnical research in recent years seems to have alternated between reliance on in-situ test results and improved correlation factors for ex-situ tests. Many geotechnical soil properties are correlated to various common in-situ testing procedures such as the standard penetration test (SPT), the cone penetrometer test (CPT), etc. (Robertson and Ghionna, 1987). More technologically innovative procedures involving geophysical tests are popular as well. The limitations of these tests are that absolute values for the properties under investigation are seldom supplied. Usually, they provide relative value distributions that must be coupled with physical coring or testing results at discrete points within the sampled profile.

2.2.4 Hydrogeologic Properties

The hydrogeologic properties of a soil might be epitomized in the form of its hydraulic conductivity. That is, the specific flux of ground water for a given soil under a unit hydraulic gradient. This property is often described as being comprised of components related to the soil matrix (porosity, pore size and shape distributions, grain surface properties) and the fluid (viscosity and density). Those properties related to the solid matrix and independent of fluid are often grouped as the intrinsic permeability of the soil. This intrinsic permeability can be easily seen as having direct relationship to the depositional and weathering processes that formed the soil matrix. It might also be seen as having some parallels to the geotechnical properties mentioned earlier, and indeed it has historically been correlated with them. A good example of this would be the Hazen equation which relates the permeability of a soil to the D_{10} value of its grain size distribution (Hazen, 1911). With definition of the intrinsic permeability, the hydrogeologic properties' influence on other fluids, such as NAPL, might be predicted.

Defining the distribution of hydraulic conductivity in the subsurface is perhaps the most important aspect of conceptualizing a site with respect to contaminant transport and withdrawal. Unfortunately, it is also the parameter which has the widest range of possible values and distributions. Though soil can vary in conductivity to a wide degree within a stratigraphic element, it is the relative changes between these elements and the geometry of the elements' placement that are important for dispersive effects in contaminant transport. Also, very large differences in permeability between adjacent elements can lead to changes in the governing transport processes from advection and dispersion to diffusion. This problem of dual porosity will be discussed later under attenuating processes.

There are numerous methods available for determining the hydraulic conductivity distribution within a site. Direct measurement can be accomplished through the solution of Darcy's Law, as incorporated into an analytical equation modelling field measured, field created or laboratory controlled flow regimes. Techniques that apply this methodology are: aquifer tests, piezometer tests (slug tests), borehole flowmeter tests, packer tests, ring-in filtrometer tests, and core permeameter tests. As alluded to previously, hydraulic conductivity values can also be inferred from physical parameters of the soil such as: grain-size analyses, soil textural classifications, geologic facies identification and mapping, and finally surface and borehole geophysics (Gelhar, 1993). For relatively homogeneous fine-grained units in the subsurface, such as clay masses or lenses, laboratory tests might be necessary to determine the hydraulic conductivity in a reasonable time frame, although pore pressure dissipation behaviour during CPT tests has been used to derive K values (Smythe *et al.*, 1989).

The transfer of hydraulic conductivity values obtained through the above methods to the entire aquifer regime is limited by considerations of heterogeneity. A pump test simply gives an averaged value of the hydraulic conductivity tensor for the volume of aquifer within its zone of influence (K_{ef}), sometimes considered to be a function of the geometric mean conductivity. Dagan (1986) suggested the use of the following formulae for effective hydraulic conductivity in an aquifer volume and its associated variance:

$$K_{ef} = K_G * \left(1 + \frac{\sigma_Y^2}{6}\right) \qquad [2.8]$$

$$\sigma_\phi^2 = \frac{\sigma_Y^2 J^2 I_Y^2}{3} \qquad [2.9]$$

The longer the test, the more heterogeneities its zone of influence encounters. Gelhar (1993) stated that averaged data can be useful in obtaining statistical correlation scales smaller than the actual spacing of the observation points. For discrete samples however, this degree of resolution is not possible. Table 3.3 in the following chapter illustrates a variation of values for conductivity tests in the same wells of the same aquifer based on the type and length of testing.

2.3 FLUID PHASE PROPERTIES AND PROCESSES

The action of the fluid phase in the subsurface is very important as it is often the transport medium of contamination. Free-phase immiscible fluids might fall in this category, however all aspects of the contaminant will be included later in this discussion. We are therefore left with liquid and vapour phase water, or put more generally, pore water and air.

This section deals largely with the liquid aspect of water since the report centers on pump-and-treat remediation technology. However, vapour-phase transport might become significant when that solute re-enters the dissolved phase. Vapour extraction itself as a remediation technology is applicable to many of the volatile contaminants that are of concern. However, a discussion of vapour extraction technology falls beyond the scope of this report.

2.3.1 Climate

Climate, for the purposes of this paper, relates to the precipitation events and volumes that effect recharge of the ground water regime. Temperature might also be considered to some degree with respect to its influence on fluid properties, contaminant properties, and chemistry. Its main influence, however, is the effect temperature in the surface environment has on evaporation or volatilization rates and surface permeability, and the effect climate has on vegetation and the subsequent transpiration rates.

Precipitation of water onto the surface of the ground is one of the principal sources of water influx into the subsurface regime of a site. This encompasses both direct recharge and discharge through the surface of a site and recharge/discharge offsite which affects the regional hydrogeology. These will both be discussed following this subsection.

The distribution of precipitation both areally and temporally is usually considered random with respect to contaminant hydrogeology. The spatial variation might also be considered minor on the local scale when compared to temporal variation. This distribution encompasses not only the discrete increments of 'on' and 'off' when discussing precipitation events, it also encompasses the intensity of the water delivery. The precipitation event might again be divided into liquid phase delivery that occurs during rainfall and solid phase delivery as ice or snow that would continue to the subsurface only upon melting. Fortunately, since the time frame of contaminant transport problems is usually much larger than a single season, let alone a single precipitation event, the question of water delivery from the climate is usually handled as an average annual or seasonal rate.

Temperature in the environment also has high variation on several time scales (daily, seasonally, etc.). This individual aspect is not usually considered in contaminant transport analysis, but in the net recharge rate.

Determination of the climate variables for a given site is relatively straightforward. It involves examination of historical records of both precipitation and temperature. Inference is thus seldom necessary at this level in the investigation. Prediction of future climate values is usually simply an extension of historical seasonal values. The measurement and prediction of these variables has become a well established science that relies on the ease of measurement of these above surface processes and stochastics in characterizing their random nature.

2.3.2 Local Recharge and Discharge

Local recharge and discharge encompasses the volumetric rate as well as both areal and temporal distribution of water delivery to and from the subsurface in the area encompassing a contamination and remediation investigation. As suggested previously, these values are usually averaged over a season or a year.

The rate of recharge is dependent on many other elements of the regime. The topography and precipitation intensity affects the rate of runoff, which, combined with the soils' vertical conductivity, dictates the rate of percolation into the subsurface. Also, other climate variables affect evaporation and transpiration to provide sinks of moisture. These must be subtracted from gross percolation rates to arrive at the net recharge to the ground water regime, both locally and regionally.

Variation in patterns of topography, areal distribution of vertical hydraulic conductivity in the surface soil materials, and variation in vegetation types can lead to significant variability of the recharge rates to the subsurface across a site.

Direct determination of the recharge rate into the ground water regime has not been accomplished to any notable degree. Usually, recharge is estimated through predictive models incorporating topographic, hydrogeologic, climatic and vegetative input. Another common method of determination is through inference using detailed models of the ground water regime. Inverse modelling techniques are used to narrow the estimates of net recharge and influx through model boundaries. This will be expanded upon in a later section on inference and prediction methods (Section 2.6.4).

2.3.3 Regional Hydrogeology

Regional hydrogeology refers to the behaviour of the ground water regime in the large area surrounding a site. In terms of the contaminant transport system at the site scale, many of the recharge and discharge considerations over this larger area would be translated to subsurface influences of the ground water regime on the boundaries of a site.

The same climatic influences on the local scale would apply to the regional scale. However, the larger area involved might lend itself to areal averaging more easily when consideration is centered on its effect on a site's hydrogeology. The interface between the regional hydrogeology and site hydrogeology is largely limited to the subsurface boundaries in the saturated zone. The geology and geomorphology of the region would come into play with their direct influences on property distribution on the larger scale.

As with local scale recharge and discharge, the regional hydrogeology's effect on the local subsurface conditions of a site might be inferred through inverse modelling techniques. The expanded scale of the problem would limit the application of predictive models, though they might be used in a simplified form.

2.4 HYDROSTRATIGRAPHY AND HYDROGEOLOGY

The hydrostratigraphy of a site might be defined as the spatial relationship of materials having similar hydrogeologic properties and similar effects on ground water flow within the subsurface regime. For the purposes of this report, this will be considered to encompass similar contaminant transport properties as well.

This component of the regime has input from the hydrogeologic properties, local recharge/discharge processes and regional hydrogeology. Together these influences dictate the volumes and discharge rates of the ground water which are distributed according to the hydrogeologic properties of the soil. The ground water flow distribution is directly responsible for the physical transport of contaminants through the subsurface, along with chemical transport processes such as molecular diffusion which is driven by concentration gradients.

The variability of the hydrostratigraphy in a site's subsurface is very large. It is the product of influences from hydrogeologic properties, which are themselves of very wide variation, and flux inputs from recharge sources. The product of these elements creates variability that is comparable if not greater than the hydraulic conductivity distributions of the soil. Large differences in the flow regime between adjacent hydrostratigraphic elements can often lead alteration in the dominant transport processes from advection-dispersion to molecular diffusion.

Hydrostratigraphy and hydrogeology can be initially defined through historical sources, such as supply well data, and hydrology studies. These can be supplemented by the installation of new piezometers and monitoring wells as well as by initiating new hydrologic interpretations of historic data, perhaps with the improved modelling techniques available today.

Ground water velocities in both the natural regime and the pumped environment can be determined from historical plume migration of conservative contaminants, or calculations applying Darcy's Law to the hydraulic gradient and hydraulic conductivity distributions. The resolution of ground water velocity distribution is therefore usually limited by the resolution of hydraulic conductivity and hydraulic head distributions, if historical plume migration is not available. There are also some in-situ tests available that use tracer dilution rates as indications of the pore water velocity. These 'point dilution methods' have been applied to contamination investigations in the past (Jackson *et al.*, 1985).

Hydraulic gradients are largely determined from the distribution of hydraulic head throughout the various aquifers in the subsurface. This may be difficult to define if the complete stratigraphy, in terms of interconnectedness of hydrostratigraphic units, is not accurately known. Also, the change in hydraulic gradient distribution when pumping stresses are applied may be very complex and impossible to define with the discrete data available from observation wells. Vertical variation in gradient within an aquifer can be crucial to the understanding of migration processes as well, if the geometry of the unit promotes three dimensional effects.

Aquifer interconnectivity is a very important consideration when assessing the contamination potential of a site. In layered aquifer systems, the upper unconfined aquifer is the first to be penetrated by influx of chemicals. Ground water can be withdrawn from a variety of levels; shallow aquifers are often utilized for small private use and deeper aquifers for public or industrial supply wells. Though penetration of contaminants through intact aquitards is possible and must be considered, migration through aquitard discontinuities or fractures is probably more crucial for the time frames usually considered in contamination scenarios. Also, deep aquifers (used for public water supply) can experience such heavy utilization that enhanced downward gradients are created (Smedes *et al.*, 1993).

Pumping well tests, with multi-level response monitoring, can determine the level of hydraulic interconnectivity between the aquifers (Meiri *et al.*, 1990). Interconnected aquifers greatly complicate the characterization of contaminant transport regimes, especially with respect to DNAPL migration. Differentiation between a highly heterogeneous aquifer with numerous clay lenses and a series of interconnected aquifers is often difficult. Three-dimensional modelling is often required (with the associated increase in data requirements) to predict behaviour in these environments.

Undetected aquitard fracturing can also lead to erroneous predictions of inter-aquifer contaminant migration. The presence of fractures can drastically increase the effective hydraulic conductivity of the aquitard, above values determined in laboratory tests of small intact samples. Properly monitored pump tests can indicate this type of interconnectivity as well as aquitard discontinuities, though they may not differentiate between the two. Site investigation techniques to determine fracturing can be very expensive and of limited success due to the discrete nature of fractures.

Recent developments in geostatistics and stochastics have been applied to characterize the aquifers stochastically. This has generally been applied to estimation of macrodispersion coefficients but recently predictions of material distribution have been attempted (Fogg, 1986; Cromer and Srivastava, 1992; Leeder, 1989). Though little can be done to eliminate preferential flow channels, understanding the total hydrogeological flow regime can lead to more effective system design and more realistic project goals.

2.4.1 Physical Attenuation

Physical attenuation processes in contaminant transport and withdrawal include those associated with mechanical dispersion and molecular diffusion, together referred to as hydrodynamic dispersion. The mechanical dispersion component of attenuation is largely connected to dilution of the contaminant through mixing action in a complex flow regime. Molecular diffusion has been alluded to previously when referring to low hydraulic conductivity soil elements. It involves the transport of contaminants in response to a chemical gradient. This process is always present when concentration gradients exist. However, it has largely been neglected when advection-dispersion action was significant. In this respect, it might be considered part of physical attenuation processes since it relies on the characteristics of the flow regime to be considered.

As mentioned above, mechanical dispersion attenuation in the macroscopic sense relies directly on the configuration of the hydrostratigraphy. This diluting action might be considered an irreversible attenuation since diluted water volumes are not commonly reconcentrated. Molecular diffusion, however, relies on both the chemistry of the pore fluid and the hydrogeologic properties of the soil. It is often neglected in the high flow areas of the stratigraphy but becomes important in the lower permeability zones. Here it can cause the soil material to act as a sink or source of contaminant mass over the period of contamination and remediation. Diffusion-based attenuation might be considered reversible, or a retardation effect, since it involves removal of contaminants from the active flow regime, and subsequent reintroduction when the gradient direction is reversed.

Stochastic methods quantify macrodispersivity in terms of the soil's statistical parameters of mean, variance and correlation lengths (Dagan, 1986; Gelhar and Axeness, 1983), and have been subsequently supported by field scale applications (Sudicky, 1986; Hess *et al.*, 1992). In-situ tests have also been applied extensively to determine field scale dispersivity in aquifers. Gelhar *et al.* (1992) provided a comprehensive database of these studies, including evaluation of their levels of reliability.

As with pump tests for flow property determination, field tests to calculate dispersivity of a soil are limited to the average value for the aquifer volume encountered by the tracer during the test. This can be an important point when macrodispersivity is on a larger scale than reasonable test volumes (i.e. single well slug tests). Inverse modelling of contaminant plume development has often been used to quantify dispersion effects in the aquifer under a natural gradient. Its use for this and other parameter determinations will be discussed later in this chapter.

Molecular diffusion processes have long been considered insignificant with respect to contaminant transport. In recent years, however, many scientists and engineers have determined that it can be the governing transport process through very low hydraulic conductivity materials. The initial research into this effect was conducted to determine the effectiveness of clay liners in hazardous waste containment (Gillham *et al.*, 1984). This knowledge can also be applied to migration through continuous aquitards, as well as to contaminant mass storage capability of discontinuous fine grained stratigraphic elements, when addressing the dual porosity effect. Tailing effects in contaminant withdrawal are sometimes partially attributed to the rate limited molecular diffusion out of these mass sinks, as well as being attributed to the chemical retardation and attenuation effects described later in this paper. For fractured media, molecular diffusion may govern in extracting residual and dissolved contaminant from dead end fractures (or pores) and the host matrix, where advective flux is negligible.

Diffusion properties found in the laboratory are usually transferable to the field due to the relative homogeneity of fine grained soils. Determining the distribution of these fine grained soil units is more problematic. From these diffusion properties, diffusive transport through continuous aquitards can be quantified. Quantification of the mass storage in low hydraulic conductivity zones and dead end fractures or pores is less easily done. The rate of reversed diffusion back into flow channels and extraction systems is similarly difficult to determine. Dual media flow and transport models have shown some promise in this area (Herr *et al.*, 1989; Latinopoulos and Katsifarakis, 1991).

2.5 CONTAMINANT PROPERTIES AND PROCESSES

The contaminant itself is of central concern in the contamination remediation scenario. The nature of the various chemicals of concern can have very significant effects on the behaviour of the contamination plume both upon migration and withdrawal. This extends beyond major categories such as organic or inorganic compounds, and even beyond the subcategories of LNAPL and DNAPL compounds when discussing immiscible organic chemicals. The combination of numerous specific characteristics of the contaminant, or even differential characteristics between several contaminants, can result in a specific scenario to be dealt with upon investigation and remediation.

The contaminant influences the subsurface regime in several ways. These influences are manifested as: characteristics of the actual release of compounds into the environment; various aspects of the dissolved chemistry and the free-phase liquid chemistry; and interaction between the contaminants and the solid matrix through geochemistry. All of these components lead to the solute and free-phase contaminant distribution in the subsurface and the processes involved in moving this mass to the surface through pumping.

2.5.1 Contaminant Release

There are several types of contaminant releases dealt with through pump-and-treat remediation. These are typically: long-term slow leaks of chemical supply lines or storage facilities (i.e. chlorinated solvents at industrial sites); accidental spills involving a single, short term release from storage tanks, supply lines or transportation vehicles (i.e. freight train derailment); multiple short term release events from repeated practices (i.e. fueling spills at airports); or continuous infiltration from surface impoundments (i.e. infiltration ponds). Releases might be: pure phase contaminant of one single compound or several compounds, or contaminants already in the dissolved phase. Also, when several compounds are involved they may be of similar or completely different characteristics. Examples of each of these scenarios are presented in the following chapter on case studies in pump-and-treat remediation.

Upon interaction with the ground water, the compound released has a direct effect on the resultant pore water chemistry. Interaction between the contaminant and the soil matrix depends on the geochemical characteristics of both components. Contaminant releases can vary in volume, rate of release, and type of compounds released. The volume released into the subsurface sometimes depends on the type of release. Each type of event can result in small or large releases depending on the adequacy of preventative practices that were employed. Typical volumes, in case histories, have varied from a few cubic meters to thousands of cubic meters (see Chapter 3). Periods over which releases have occurred have varied from minutes (eg. catastrophic spills) to decades (eg. long-term site operation practises).

Accidental spills are perhaps the most straightforward to identify and to quantify due to stringent record keeping requirements under current environmental regulations (in most developed countries). Long-term industrial releases over decades are perhaps the most difficult to identify, both in terms of contaminant types involved and volumes released. The requirement of keeping inventories of hazardous chemicals on site or in transit has slowly developed over the last few decades, leaving many unknowns for operations that existed previous to this time frame.

Identification and quantification may involve examination of the release site and chemical inventories, depending on the contamination scenario. At the other extreme, it may require extensive subsurface investigation, wide spectrum chemical analyses, and interpretation of likely contamination distributions. The results may be highly uncertain even after this complex process.

2.5.2 Chemistry

The chemistry of a contaminant plume involves the aqueous concentrations of the various elements and compounds of concern, as well as other dissolved constituents in the ground water that may have an influence on the contaminants' behaviour. The chemical composition of the ground water may change over time due to various chemical reactions, mass sinks and sources, element or compound mass influx, and diluting action. For the purposes of this report, chemical reactions will also include biochemical and radioactive decay processes.

The chemistry of a volume of contaminated ground water is influenced by mass influx of various contaminants due to a release event, and mass influx of other constituents from ground water recharge and regional hydrogeology. Adsorption and release of chemicals to and from the solid matrix also has direct influence on the aqueous chemistry of the pore water.

The variability of chemistry in the ground water has certain constraints from physical laws such as the solubility of the constituent in question, which is usually dependent on several other environmental factors such as the temperature and the rest of the chemical make up of the pore fluid. Also, the exposure of each soil zone to contaminant influx due to its relative position within the transport field leads to significant concentration variations. Ronen *et al.* (1986) presented the wide variation of chemistry that can occur within a contaminant plume over very small distances, on the order of centimeters.

The first concern is identification of the contaminants to be tested. In certain regulatory regimes, or where there is a lack of information relating to the contamination source, testing of a broad range of chemicals must be performed at the initial stages of an investigation in order to fully define the nature and extent of contamination. Once elevated levels of certain chemicals, or groups of chemicals, have been identified, they are the focus of more specific sampling and testing. In the later monitoring stage (during remediation or site maintenance) fewer 'indicator' chemicals might be chosen to track the progress of withdrawal with full spectrum testing reinstated for closure action. Istok *et al.* (1993) discussed the statistical reliability of using concentrations of a more easily measured substance to delineate the concentration of another associated substance. This will be discussed later.

Gravity driven flow of NAPL downward through the vadose zone (and through the saturated zone in the case of DNAPL) leaves behind residual contamination, adsorbed contamination and free-phase pools over NAPL-impervious layers. While residual and adsorbed NAPL can be located and quantified by core samples through the migration zone, pools of free product can be difficult to locate and very difficult to quantify in complex heterogeneous regimes (Kueper *et al.*, 1989). Fluid concentrations that are found to be greater than the contaminant's solubility can indicate the presence of free product in the extracted fluid volume. Also, a direct measurement of saturation level, and comparison to residual saturation for the soil structure, can aid in locating pooled NAPL (Murarka *et al.*, 1992).

2.5.3 Geochemistry

The geochemistry of a soil with respect to contaminant transport refers to its properties of chemical interaction with the pore water. This includes such soil properties as: cation exchange capacity (CEC), specific surface area, adsorption capacity and behaviour (isotherm characterization), organic carbon fraction (f_{oc}), chemical reactivity of soil material (mineralogy), ion exclusion zone width, wetting angle of liquid phase contaminants, etc. These properties can have negligible to significant effects on the fluid chemistry during transport, depending on the full details of the regime.

Some of the above properties derive from the soil grain mineralogy, such as the CEC, chemical reactivity, ion exclusion zone width. Others have more direct relationship to the weathering and depositional environment, such as the organic carbon fraction, specific surface area, and wetting angle. The adsorptive capacity of the soil is a combination of a few of the above factors. The value or nature of the geochemical properties of the soil is also very specific for each contaminant or chemical species.

The variability of the geochemical properties of the soil is often parallel to the stratigraphic units of the soil profile. That is, those elements of the subsurface which have experienced common depositional or weathering environment and source material. However, variation within the element is also possible as the distribution of these materials changes. For example, post-depositional weathering may have significantly changed the geochemical properties of the near surface portion of a stratigraphic element while the lower portions remained unchanged.

Laboratory testing techniques can determine the various geochemical properties of soil: including, pore water chemistry, organic carbon fraction (f_{oc}), cation exchange capacity (CEC), specific surface area, adsorption capacity and behaviour (isotherm characterization), chemical reactivity of soil material (mineralogy), etc. However, in-situ tests for many of these properties have historically been limited in availability and practical application. Transfer of laboratory results to field regimes must usually be relied upon for geochemical characterization of a site.

Some advances have been made in the area of in-situ geochemical testing, however. Field testing labs (mobile or on site) have become more common and economical. The methodology of site investigation which incorporates encountered data into continuing investigation patterns dictates that analysis response time be minimized. This practice is exemplified in site investigation case histories such as that described by Murarka *et al.* (1992) and in field analysis overviews such as that illustrated by Chudyk (1989).

The mineralogy of the various soil facies within the subsurface can be very helpful in predicting the geochemical processes that will occur during contaminant transport and withdrawal. The determination of soil mineralogy can be very expensive and time consuming and must be limited to those areas where it will yield the most useful information. Clay fractions are commonly the most chemically reactive soil components. Consequently, understanding the mineralogy of this fraction is more important than for other elements of the soil. The most common procedure for determination of the clay mineralogy is through laboratory testing of borehole samples.

Along with the clay component, organic carbon is another highly geochemically active fraction of the soil. Where clay largely affects the ionic (commonly inorganic) contaminants, organic carbon has its effect on the nonionic (commonly organic) contaminants that are of steadily increasing concern.

2.5.4 Solute Transport

Solute transport is the migration of dissolved contaminant compounds through the ground water regime. This is accomplished through a combination of advection, dispersion, and diffusion processes. The dominant process depends on the nature of the hydrostratigraphy and the chemistry in the soil environment. Commonly, high permeability homogeneous material results in advection dominated transport, high permeability heterogeneous material results in dispersion dominated transport, and very low permeability material leads to diffusion dominated transport.

The processes of advection and dispersion are often linked, since present definitions of these processes couple them through the advection-dispersion equation. Diffusion acts in all groundwater environments where concentration gradients exist, although its effect is often negligible unless the ground water velocities in the direction of interest are very low. In high flow regimes, with very low lateral dispersion properties, diffusion may govern contaminant transport normal to the flow direction.

As suggested by the above description, solute transport is very heavily influenced by the contaminant transport components of hydrostratigraphy, hydrogeology, chemistry, and geochemistry. The hydrogeology's influence is shaped by advection and the physically attenuating process of dispersion. Molecular diffusion might also be included in this category, though it has a chemistry basis that has been described previously. Both chemistry and geochemistry influence the transport of dissolved constituents through chemical retardation and attenuation, to be described later.

Solute transport processes acting on a contaminant plume are often determined through inference. Chemical delineation of the plume over time is taken as input to derive the operative processes. These processes are balanced in their relative degree of influence to match the observed pattern of the contaminant plume, resulting in identification and valuation of the various parameters involved in contaminant transport. A later section on modelling, specifically with respect to inverse modelling, will discuss this in further detail. Prediction of solute transport at later times can be accomplished once the operating parameters have been determined and calibrated (Peck *et al.*, 1988).

2.5.5 NAPL Transport

The transport of nonaqueous-phase liquids (NAPL) through porous media involves many of the variables associated with ground water flow, though the parameter values may be quite different. Migration of free-phase immiscible fluids through the unsaturated and saturated zones of the subsurface is essentially two (or sometimes three) phase flow. It involves pressure gradient induced flow and gravity driven percolation, as well as capillary-force driven migration. The wetting curves of the NAPL in the water-soil or air-water-soil environments are parallel to the moisture retention curves of water and air in unsaturated soil.

The migration behaviour of NAPL through the subsurface is dependent on several aspects of the solid matrix that relate closely to its hydrogeologic properties. As mentioned previously, the intrinsic permeability of a soil can be used to characterize its behaviour under free-phase NAPL flow. The pore geometry of the soil also affects the NAPL behaviour through the retention characteristics mentioned above. Finally, the adsorption affinity and wetting characteristics of the immiscible fluid to the solid matrix would directly affect its behaviour.

Fractures or discontinuities through NAPL impermeable stratigraphic elements (commonly clay-rich aquitards or lenses) are very important to the mobility of the immiscible fluid. The difficulty in determining the distribution of these elements adds to the complexity of characterizing the physical processes of NAPL migration through porous media. The problems associated with investigating aquifer interconnectivity have been discussed previously with respect to hydrogeologic characterization.

There is always the risk that extensive site characterization, through drilling or cone penetrometer tests, might influence the very properties being measured. This is especially true for tests of hydraulic interconnectivity in lenticular deposits. The borehole may itself provide a channel for cross contamination of previously isolated aquifers. There is also the risk that perched water-tables or NAPL trapped on aquitards (of limited or continuous extent) may have their hydraulic isolation breached by a poorly installed borehole. This process could be thought of as a hydrogeologic application of the Heisenberg uncertainty principle, where the process or property under observation is effected by the very act of observation.

Definitions of relative permeability, displacement pressures, critical pool height and saturation characteristic curves for multiphase fluids in porous media are largely accomplished in the laboratory. Distribution of these properties through the vadose zone and aquifer might be accomplished using soil samples of grain size distribution and residual NAPL content (Mercer and Cohen 1990).

2.5.6 Chemical Retardation and Attenuation

Chemical retardation involves reversible processes that act to slow the contaminant's migration rate through the soil matrix. These processes include partitioning between the solid matrix and liquid phases of the soil regime and precipitation of constituents into a solid phase. Attenuation of contaminants refers to irreversible processes which reduce the concentration in the aqueous phase, including such processes as: biochemical degradation, chemical complexation, and radioactive decay. Both types of chemical action have significant effects on the transport of contaminants. It is important to note that the products of attenuation should be considered as possible contaminants as well.

Chemical retardation and attenuation affects the interactions between chemistry, geochemistry, and solute transport components of contaminant transport. It also affects the NAPL transport to some degree and the subsequent interaction between it and solute transport.

Adsorption properties of the soil material for various contaminants has been historically determined in the laboratory. A well defined procedure or 'batch test' to quantify this property has been standardized by ASTM (ASTM D4319-83 and D4646-87). However, there is some question as to the transfer of batch test adsorption (isotherm) results to field behaviour. Column tests have also commonly been used, and are generally accepted as more easily translated to field behaviour.

Adsorption is often a process affecting organic chemicals as they react with organic carbon fractions of the soil. Transfer of the laboratory determined octanol-water partition coefficient (K_{ow}) values for the contaminant to field distribution coefficient (K_d) values is a common practice. Assumptions that K_d is dominated by molecular adsorption to the organic carbon fraction usually hold for $_{oc}$ greater than 0.1% (Jackson *et al.*, 1985).

Sometimes these geochemical and biochemical effects are recognized but are lumped into one or two general parameters (i.e. retardation coefficient and decay coefficient) and defined through inverse modelling. Detailed coupling of physical transport and chemical reaction processes has not been accomplished to any great degree at present. This will be discussed further in a later section on modelling.

Present-day analysis methods usually utilize numerical or analytical models to accommodate complex design problems. However, the data requirements of such models are often too onerous for most reasonable remediation projects without numerous simplifying assumptions. Some examples of these simplifications are:

1) contaminant flow is sufficiently slow to allow equilibrium chemistry to be obtained;

2) adsorption and ion exchange effects are instantaneous and reversible; and

3) individual species in multiple solute plumes do not interfere with each other.

Without these assumptions, inverse modelling techniques often cannot obtain unique solutions without prohibitive data requirements (Peck *et al.*, 1988).

Non-ideal effects in contaminant transport are often neglected in favour of (relative) simplicity in conceptualization. Brusseau (1992) discussed possible deviations from the common linear, instantaneously-reversible sorption assumption, including: non-linearity, irreversibility, non-equilibrium, and capacity variability. Brusseau (1992) also mentioned facilitated transport and hydraulic conductivity variability as other possible deviations. Obviously, homogeneity with respect to hydraulic conductivity is no longer a common assumption in most applied cases. Facilitated transport due to solvent or colloid action in reducing retardation and attenuation is of growing interest. Several other recent papers have reviewed possible non-ideal effects in contaminant withdrawal with respect to pump-and-treat remediation schemes (Haley *et al.*, 1991; Hall and Johnson, 1992; Hockman, 1992), including heterogeneity of hydraulic parameters.

Physical distribution of soils with differing geochemical properties, such as adsorption or ion exchange capacity, can also lead to increased macrodispersion by varying the contaminant velocity, even though the ground water velocity remains relatively constant

(Chrysikopoulos *et al.*, 1990). If the variation in retardation properties follows variation in hydraulic conductivity, the macrodispersion can be further exaggerated. This can often be the case, since finer grained soils often have higher surface area and higher clay or organic content, as well as lower permeability (Barber II *et al.*, 1992).

2.6 REMEDIATION INVESTIGATION

Remediation through pump-and-treat technology is one of the few techniques that has been applied with regularity to large scale contamination problems. With respect to the flowchart of contaminant hydrogeology in Figure 2.1, pumping action might be considered to alter the local recharge/discharge component. This subsequently changes the relationships between many of the other components in its implementation, most particularly the hydrostratigraphy and hydrogeology and, upon mass withdrawal, the chemistry.

The various elements of a pump-and-treat remediation investigation are: site investigation of those characteristics which can be directly determined, interim action to alleviate the magnitude of future remediation, monitoring of both physical and chemical aspects of the site, and inference or prediction techniques.

2.6.1 Site Investigation

Site investigation involves the initial determination of solid matrix, fluid phase, and contaminant properties and their distribution across a site. This applies various new and established technologies to delineate the various components of Figure 2.1. Some of these elements have been more traditionally investigated, such as hydrostratigraphy, hydrogeology and chemistry. Others have become important as parts of a contamination investigation, such as geochemistry and NAPL distribution.

The general methodology for site investigation of remediation projects differs somewhat from traditional geotechnical or geological investigations. Though all engineering works require consideration of legal aspects in their implementation, the present litigious atmosphere toward environmental issues, particularly in the U. S., makes observation of strict procedures especially important. Burgess (1991) provided a detailed outline of investigation of contaminated sites that pays particular attention to quality assurance/quality control of borehole installation, well materials, sample collection and analysis in this area.

Statistical analysis techniques that have been discussed previously in this report can be applied to the many variables to be determined in a site investigation. The number of samples required for a given confidence level in the mean or variance of these properties can be quantified using stochastic theory. This quantity of samples is dependent on the various statistical parameters which characterize the property in question. This process has been discussed in more detail previously in Section 2.1.2.

The initial characterization of a site is usually based on available information obtained without initiating a new investigation. This characterization can be extensive or practically non-existent, depending on the amount of previous work done on or near the site (Edwards *et al.*, 1983; Burgess, 1991). Government agencies at several levels often maintain geological and hydrogeological information about the land under their jurisdiction. The level of detail of this information often depends on the site's proximity to other development (private, commercial or industrial).

2.6.2 Interim Action

The data collected in the first stages of site investigation are likely sufficient to design some temporary containment system. This containment may be hydraulic or physical. Quick removal of heavily contaminated soil or source material, and possible reduction of infiltration through known contaminated material volumes, coupled with prompt response action, can drastically reduce the extent of contamination to be removed by the full scale remediation project.

There are some negative aspects to the practice of quick interim action, however. If the interim system is applied too soon in the investigation some valuable data may be lost. This includes natural ground water regime information and contaminant delineation that could be used in applying inverse modelling techniques to refine flow and contaminant transport parameter estimates. It is therefore necessary to balance the benefits of early interim action against the value of data lost by its implementation (Haley *et al.*, 1991). The difficulties in inference efforts that can result from lack of 'natural' flow regime data are illustrated in the modelling case study of Chapter 4.

2.6.3 Monitoring

Monitoring is perhaps the most important post-design aspect in any ongoing remediation project. Since the majority of remediation action occurs in the subterranean environment, almost none of the responses are apparent through surface observation. Further, chemical responses must be assessed through field unit or laboratory analysis in all but the most extreme cases. Management toward optimal removal cannot be achieved without detailed, accurate data on the hydrogeological and chemical state of the subsurface, as well as its response to the remediation system operation.

Many variables exist in the realm of remediation monitoring. Several aspects of the monitoring program's physical configuration must be defined, including: where to locate monitoring wells (especially in highly heterogeneous media), how many monitoring wells to install, how often to sample these wells, how much effect the monitoring has on the property it monitors, and is the monitoring process itself increasing the contamination risk. There are also several decisions to make regarding the chemical testing program, such as: what substances to test for, what detection levels are reasonable, what confidence can be assigned to given data, etc. Finally, data requirements must always be balanced against economic constraints. All these concerns must be addressed in order to maintain an efficient and effective remediation project.

2.6.3.1 Physical Monitoring Considerations

Locating monitoring and remediation wells is often an iterative process; each newly installed well provides more information on the site's geology, hydrogeology, geochemistry and chemistry. This information may subsequently dictate a change in the previously planned borehole/well layout. As is implied by this statement, much of the location and installation of wells occurs during the site investigation stage. Burgess (1989, pp.3147) lists the following objectives of monitoring well installation:

- "define geological stratigraphy
- obtain soil samples for physical and/or geotechnical testing
- determine piezometric head
- perform in-situ hydrologic tests
- obtain soil samples for chemical analyses
- obtain water samples for chemical analyses
- obtain samples of separate phase fluids, for example floating petroleum product or dense chlorinated solvent
- obtain samples for characterization of microbial population."

Previous sections have described the solid matrix, fluid, and contaminant properties that should be delineated during investigation. The boreholes made during this operation are often subsequently used for both monitoring well and extraction well components of the remediation system. These installations may be supplemented to complete the requirements of the design according to various data needs and optimal spacing or density of wells.

The density of monitoring or remediation wells is very subjective. Heterogeneity of a site, budget constraints and the contaminants' risk level will largely dictate the density of the well layout and the scope of analyses. It is important to remember the regulatory environment at this point. Some regulatory regimes dictate installed wells be maintained and monitored in perpetuity, which would have a very serious impact on the project's long-term operating budget. To address these concerns, studies have been undertaken to optimize monitoring well allocation using stochastic analysis and optimization theory (eg. Morisawa and Inoue, 1991).

The location of these wells is an even more complex issue. It is influenced by a number of variables: the quantity of wells to be installed (discussed above), the nature of the subsurface environment (both geologically and hydrogeologically), the extent of the contamination, the toxicity of the contaminant in question, and the proximity of sensitive facilities, environments or properties (drinking water supply wells, surface water contacts, residential areas, etc.).

Monitoring of the ground water hydrology, in both its natural state and in response to pumping induced gradients, is a well understood field. Various types of piezometers (open, closed and valved) have been applied to geotechnical works for several decades. Black *et al* (1989) provided a good overview of the state of the industry in piezometer use. Advances in electronic monitoring of ground water levels have led to the application of remote monitoring techniques (Bordes *et al.*, 1991) that may be useful in application to long-term remediation or containment projects.

There appears to be some incompatibility in suggested installation practices for chemical sampling or ground water monitoring. Burgess (1991) for instance, stated that only single well completions per borehole be installed to avoid risk of cross contamination. However, Smith *et al.* (1991), Korte and Kearl (1991), Ronen *et al.* (1986), and others identified the need for very fine vertical delineation of contamination in the ground water. A nest of single completion wells to obtain this resolution would most likely disturb the regime beyond the ability of interpretation methods to rectify, especially in areas of thick saturated zones.

2.6.3.2 Chemical Monitoring Considerations

Extremely variable geochemistry makes accurate plume monitoring very difficult (see Barber II *et al.*, 1992; Ronen *et al.*, 1987; Back and Baedecker, 1989). Applying stochastic theory to plume delineation may help construct meaningful conceptualization of the subsurface chemical distribution (Woldt *et al.*, 1992).

Burgess (1991) listed the US five level data quality categories as follows:

•	Level 1	Screening
•	Level 2	Field analysis
•	Level 3	Standard analysis
•	Level 4	Confirmational
•	Level 5	Non-standard

This categorization of analysis level can be very helpful in obtaining a cost effective delineation of site contamination. Although careful engineering judgment should be used in allocating the usage of these levels at various stages.

One very important issue to address, when considering the chemical data acquired through monitoring, sampling and subsequent analysis, is the confidence that can be placed in this data. The precise protocol to be followed in chemical sampling becomes more important as the level of analysis (as described by Burgess (1991)) rises. Recent advances in chemical testing abilities have made low parts per billion concentrations measurable for many compounds of concern. In fact, statistical methods have been applied to sample analysis in order to derive some concentration value from a sample set where many individual samples are below detection limits (Christman, 1991). However, regulatory acceptability must always be considered when using such innovative techniques.

The critical question now becomes, is the protocol followed in sampling sufficiently rigorous to justify confidence in concentration readings at this low level? There are many variables that intrude on the sampling process. The well installation could add contamination through the drilling mud, cleaning solvents used on the equipment, aquifer cross contamination, etc. The flushing protocol before sampling must be followed consistently each time even though it might appear excessive to the field personnel. The laboratory results must be solidly linked with details of the sampling event in order to assign confidence to the individual data point. 'Chain-of-custody' forms should be applied to insure accurate knowledge of each sample's history (Burgess, 1991). Consistency of installation, sampling and analysis procedure can negate many of the uncertainties associated with individual details. The most important information is often the relative readings with respect to space or time. A rigorous procedure should enable most of the variables to be removed as a baseline reading.

The preceding discussion illustrates the need for understanding of the true validity of obtained data in remediation monitoring. The fine vertical variations of chemical concentrations that can occur in a soil regime are demonstrated by Ronen *et al.* (1987). The validity of point samples must therefore be questioned when considered independently. Not only should the accuracy of the sample and analysis be considered, but also the degree to which the sample represents the true state of the subsurface regime might be questioned as well.

There is always the question of how much of the contaminant reduction in the aquifer can be attributed to the pump-and-treat remedial action. That is, how much concentration reduction is due to other processes than withdrawal, such as chemical or biological degradation, enhanced adsorption due to the change in flow regime, etc. In the case of degradation, the toxicity of the decay products must be considered and they may need to be tested for as well. In Herkert and Aslam (1990) the initial contaminant, trichloroethene (TCE), degraded eventually to vinyl chloride which may have introduced higher risk than the original contaminant due to its higher mobility, volatility and toxicity. An increase in adsorbed mass may decrease aqueous concentrations but it does nothing to remove the risk of ground water contamination upon later dissolution. Natural biodegradation is often credited with some of the contaminant reduction, but quantitative measurement of this activity can be suspect due to incomplete understanding of the chemical and biochemical processes.

2.6.4 Inference and Prediction

The preceding sections of this chapter have underlined the degree of physical and chemical complexity in any ground water contamination problem, especially with respect to remediation. Due to this complexity, straightforward solutions are often not possible. Data acquired through investigation, monitoring and sampling cannot be immediately applied to a simple equation to assess the ground water or contaminant behaviour. It is necessary to model the environment using many interacting and sometimes interdependent variables and governing equations. Analytical models can often accomplish this task, in the more homogeneous and simplified problems. However, numerical models (using finite element or finite difference methods) are usually necessary where heterogeneity and/or other complexities make the simplifying assumptions included in most analytical models unacceptable. Indeed, the complexity of most ground water contamination scenarios and the limitations of the available data dictates that numerous assumptions are still required in the most detailed numerical models. The linking of physical and chemical processes in these models is still very rare because it is difficult to accomplish, both in terms of data requirements and computing costs.

2.6.4.1 Boundary Value Problems

Boundary value problems are used to conceptualize mathematical models for flow and transport. Freeze and Cherry (1979) defined six requirements for boundary value problems in ground water flow, though they can also apply to contaminant transport:

1) the size and shape of the flow region,
2) equations of flow *(and transport)* within the region,
3) boundary conditions enclosing the region,
4) initial conditions within the region,
5) spatial distribution of hydrogeologic parameters that control flow *(and contaminant transport)*, and
6) a mathematical method of solution.

The governing equations used for ground water flow and contaminant transport that apply to the second item in this list might be written in their general forms as follows:

$$\frac{\partial}{\partial x_i}\left(K_{ij}\frac{\partial h}{\partial x_j}\right)=S_s\frac{\partial h}{\partial t} \qquad [2.10]$$

$$\frac{\partial c}{\partial t}+\frac{v_i}{R}\left(\frac{\partial c}{\partial x_i}\right)-\frac{\partial}{\partial x_i}\left(\frac{D_{ij}}{R}\frac{\partial c}{\partial x_j}\right)=\pm\frac{G}{\theta} \qquad [2.11]$$

The assumptions used in the derivation of the ground water flow and advection-dispersion equation (equations 2.10 and 2.11 respectively) are described in detail in Fetter (1993). It is important to note the material parameters (item 5 in the previous list) included in these equations. Hydraulic conductivity (K_{ij}), storativity (S_s), retardation coefficient (R), and hydrodynamic dispersion coefficient (D_{ij}) all require defined values throughout the model's domain. Also, initial distributions (item 4) of hydraulic head (h), and concentration (c) must be known for transient analyses. Boundary conditions applying to these equations (item 3) also need to be defined.

2.6.4.2 Model Input Uncertainty

It can be seen from the six requirements listed above that uncertainty can influence the model results in several ways. There is uncertainty in the various inputs from the site investigation including: ground water regime geometry, initial conditions of the dependent variables (h or c), and distribution of the various hydrogeologic and transport (independent) parameters. There is also uncertainty in the various aspects of the modelling system: inaccuracies in the governing equations (due to simplifying assumptions), inaccuracies in the boundary conditions used, and errors arising from the analytical or numerical solution techniques applied to the problem.

Woldt *et al.* (1992) presented an investigation of the effect of uncertainty in both the hydraulic parameters of the aquifer and the initial conditions (with respect to contaminant distribution). Using stochastic realization generators for initial distributions of contaminant and hydraulic properties as input into a 3-dimensional analytical solute transport model, probabilities of contaminant levels at specified locations were determined. It was found in this study that uncertainty in initial conditions of the contaminant plume had more effect on output uncertainty than did uncertainty in aquifer parameters. This conclusion could have great significance in the application of stochastic modelling to ground water contamination and withdrawal problems. Presently, the limited application of stochastic theory to modelling has centered around aquifer parameter distributions.

One common uncertainty in model boundary conditions is the recharge flux into the aquifer. This is especially problematic since it can vary dramatically, not only in space, but in time. Steady-state assumptions for the flow model are often used since the transport modelling period often extends over many seasons or years. With these steady-state flow assumptions comes time averaged recharge values. Consequently, water-table fluctuations are ignored and their effect on the spreading and dilution of the contaminant plume are incorporated into the dispersion coefficients. Inverse modelling techniques have often been applied to narrow the ranges of estimated recharge from hydrological studies.

The variability and difficulties in measurement of material parameters and dependent-variable values have been discussed in this chapter. Errors proceeding from uncertainty in input data and inaccuracies in model configuration indicate the need for further characterization (and minimization) of uncertainty, error minimization, or a combination of the two. Errors in parameter determination can be minimized through improved technology, improved interpretation, additional measurements, and extending data collection periods (Peck *et al.*, 1988). Also, inverse modelling techniques might be used for model calibration to reduce macroscopic inaccuracies in parameter distributions. However, the possibility of the solution to an inverse modelling effort not being unique must be addressed. Uncertainty can be reduced in the same manner. Quantification of uncertainty can be accomplished through sensitivity analysis (in deterministic models) or stochastic modelling techniques.

Data requirements of ground water flow and contaminant transport models can become unrealistically large. Peck *et al.* (1988) illustrated a case of a rectangular aquifer with complex boundary conditions under transient conditions. With isotropic properties and negligible storage change assumed, a proposed finite difference model required almost 38,000 pieces of data. At this point, the data requirements were dominated by aquifer properties. Further simplifying assumptions of zoned material properties were applied to reduce the model requirements. Using spatial interpolation methods (i.e. Kriging) to distribute material properties, the proposed number of data pieces were reduced to 4,650. They noted that the data requirements were then dominated by system stresses and initial conditions, and that these could be more easily determined than aquifer parameters. These examples indicate the relative data requirements of the two models; the discretization would obviously have a direct effect on the absolute values given.

Contaminant transport modelling was also discussed in the above paper with respect to its additional data requirements. Single values for these parameters were mentioned for application to the proposed model. Discussion of soil property variability at the beginning of this section showed that static properties (i.e. porosity, clay fraction, organic carbon fraction) evidenced much less variation than the dynamic properties governing flow (i.e. K_{sat}). However, these variations may still be significant in predicting levels for contaminants that are closely regulated.

2.6.4.3 Inverse Modelling

One common solution to the extensive data requirements of numerical models applied to ground water contamination is the use of inverse modelling techniques. In this way, effective values for the various material parameters might be refined beyond initially available intervals of confidence using values of observed data that were more easily obtained (i.e. h or c). Limited data available for a site might be initially applied to the model in order to obtain a solution. The model could then be refined in order to more closely match observed data values (h or c) at discrete points. This might be accomplished through both direct and indirect methods.

Direct methods treat the observed data as independent variables and the material parameters as dependent variables of the governing equation. Solutions using this method have been shown to be quick and easy, but are often highly unstable with respect to uncertainty in the observed data. Indirect methods, conversely, treat the head or concentration values as dependent and solve the governing equations to minimize the residual (the difference between measured and calculated values) at observation points. This does not suffer from the same instabilities, but does require large computational effort. Also, depending on the extent and accuracy of available input data, the model solution may have several refined data sets that give an equally close match to observed values (Peck *et al.*, 1988). However, a narrowed selection of solution sets might be attainable, to be evaluated through further directed investigations or site responses until one most likely solution remains.

The multitude of chemical, geochemical and biochemical processes involved in contaminant transport (i.e. adsorption/desorption, precipitation/dissolution, liquid-liquid partitioning, biochemical and radioactive decay, etc.) often result in tailing effects that retard or attenuate the withdrawal of the contaminant plume. These actions are usually not accounted for in present day predictive or inverse models. Many of these contaminant transport properties might be grouped into a few 'lumped' parameters (i.e. effective dispersion coefficient, retardation coefficient, and decay coefficient). Also, modelling techniques often dictate, through simplifying assumptions in the governing equations, that many effects be neglected or grouped with other dominant process parameters. Through contaminant transport theory, it can be seen that the effective dispersion coefficient contains elements of pore scale dispersion, macroscopic dispersion, and molecular diffusion. Another example might be the lumping of adsorption, absorption, ion exchange, and precipitation into one distribution parameter (K_d); similarly, chemical complexation, biochemical and radioactive decay have been joined to become one decay parameter (λ).

2.6.4.4 Predictive or Design Optimization Models

The inverse techniques discussed above might be regarded as calibration methods for a site's hydrogeological and/or contaminant transport model(s). Once the model has been matched as closely as possible to the existing regime, it might be used to predict future migration in response to possible remediation actions. This includes the risks to receptors for no-action options, and the predicted response to various remediation (or containment) schemes.

Predictive modelling of remediation schemes involving pumping wells could be used to optimize the system with respect to a desired parameter. This may involve the minimization of the extracted volume, the extraction rate, or the operation duration. Gailey and Gorelick (1993) applied this methodology to a proposed remedial action at the Gloucester Landfill. They incorporated stochastic theory into the problem to quantify overdesign values for confidence in goal achievement. Satkin and Bedient (1988) evaluated various multiple well configurations with respect to a variety of hydrogeologic site conditions (available drawdown, natural hydraulic gradient, and dispersivity). The various scenarios arising from these variables were modelled deterministically for predicted clean-up time. Uncertainty in the various inputs was not incorporated into this study. Minimization of hydraulic gradients, extraction rate, and extracted volume were considered in a multi-objective comparison, using optimization modelling (Peralta and Ward, 1991).

2.6.5 Design and Operation

The preceding sections have described the many aspects of defining the contamination problem for design of an appropriate remediation system. However, many new complexities and problems arise when the actual remediation system begins operation. The high gradients involved in pumping might change some of the properties determined for the natural gradient regime. Full scale operation may reveal inaccuracies in parameters determined at discrete points or finite volumes. The retardation and attenuation processes described in a previous section continue and perhaps expand when the flow field is reversed or altered. Heterogeneity of the soil environment could result in unpredicted responses to the pumping system, no matter how detailed the preliminary work. In some cases the pumping system might be limited by less than optimal physical conditions such as shallow aquifers (with subsequent limited drawdown available) and high dispersive fines content or biomass that could lead to well clogging. Finally, many of the predictable results of the remediation scheme could still leave large difficulties to be overcome.

Heterogeneity in aquifers has been discussed previously. The varied flow regime that results from this heterogeneity has its own effect on the distribution of the contaminant through macrodispersive processes and flow channelling. When the increased flow rates and changed flow vectors of a pumped system are applied to the regime, these difficulties could often be magnified. Channeling of flow through preferred pathways could have a negative effect on the rate of contaminant mass withdrawal, especially if this set of paths is opposed to the natural gradient flow paths. Migration into various channels might not be totally reversed by the induced gradient of the pumping well. This would be especially true of fractured rock aquifers.

2.7 SUMMARY

The preceding chapter has outlined the complexity of the contaminant transport regime, especially with respect to pump-and-treat remediation. The various elements of this regime, illustrated in the flowchart of Figure 2.1, have been described in general terms and categorized as solid matrix, fluid, or contaminant related properties, along with their interactions and influences. Aspects of their variability and the subsequent effect of sampling size was then presented. A discussion followed this that defined the various elements of the regime in more detail. This discussion included description of the component and its characteristics. General methods of determination were also introduced.

The remediation investigation process was described with respect to site investigation, interim action, and sampling or monitoring. Particular attention was paid to the value of monitoring data when both physical and chemical effects were considered. Modelling methods, as used for inference and prediction actions, were then discussed in detail. The components of the boundary value problems used for these methods were reviewed and the effects of input data uncertainty on modelling results discussed. Inference was introduced as a method to reduce modelling errors that did not exert excessive demands on the investigation process. Also, predictive modelling was presented as a procedure for optimizing pump-and-treat design configurations. Finally, the further difficulties that might appear upon full scale system operation were briefly introduced for consideration.

Full delineation of the properties discussed in this chapter might not be necessary for adequate definition of the contamination and remediation problem. Increased definition of the more critical properties could be achieved through grouping insignificant or irrelevant parameters into a few lumped values. Sensitivity analysis or stochastic modelling might then be used to determine the sensitivity of the regime to changes in those neglected properties.

Local recharge/discharge and solute transport could be considered the central components of Figure 2.1 with respect to pump-and-treat remediation. The local recharge/discharge element would be significantly altered by the introduction of extraction (and possibly injection) wells. This would result in a change in the hydrogeology and a subsequent change in the solute transport patterns, preferably toward extraction points. However, this alteration of the local recharge/discharge element does not act on hydrogeology and solute transport alone. Changes in other connected components would also be important and might act to impair the projected efficiency of this change in regime. Therefore, it would be important that all the interrelationships be understood at least qualitatively, when designing and operating a pump-and-treat remediation system.

Table 2.1: Coefficients of Variability for Various Soil and Aquifer Properties

	Parameter	Number of Studies	F: 10%	50%	100%	Average CV ± S_D
			Number of Samples.			
1	Bulk density or porosity	13	4	*	*	10 ± 6
2	Percent sand or clay	10	30	1	*	28 ± 18
3	0.1 bar water content	4	8	*	*	14 ± 7
4	15 bar water content	5	24	1	*	25 ± 14
5	pH	4	2	*	*	8 ± 3
6	Saturated K (soil)	13	591	24	6	124 ± 71
7	Saturated K (aquifer)	13	152	6	2	63
8	Infiltration rate	8	157	6	2	64 ± 26
9	K_0 in $K(\theta)$	4	1180	47	12	175 ± 139
10	Ponded solute velocity	1	1450	58	14	194
11	Unsaturated solute velocity	5	148	6	1	62 ± 9
12	Solute concentration	4	138-649	6-26	1-6	60-130 (range)

* Formula gives 1 or less

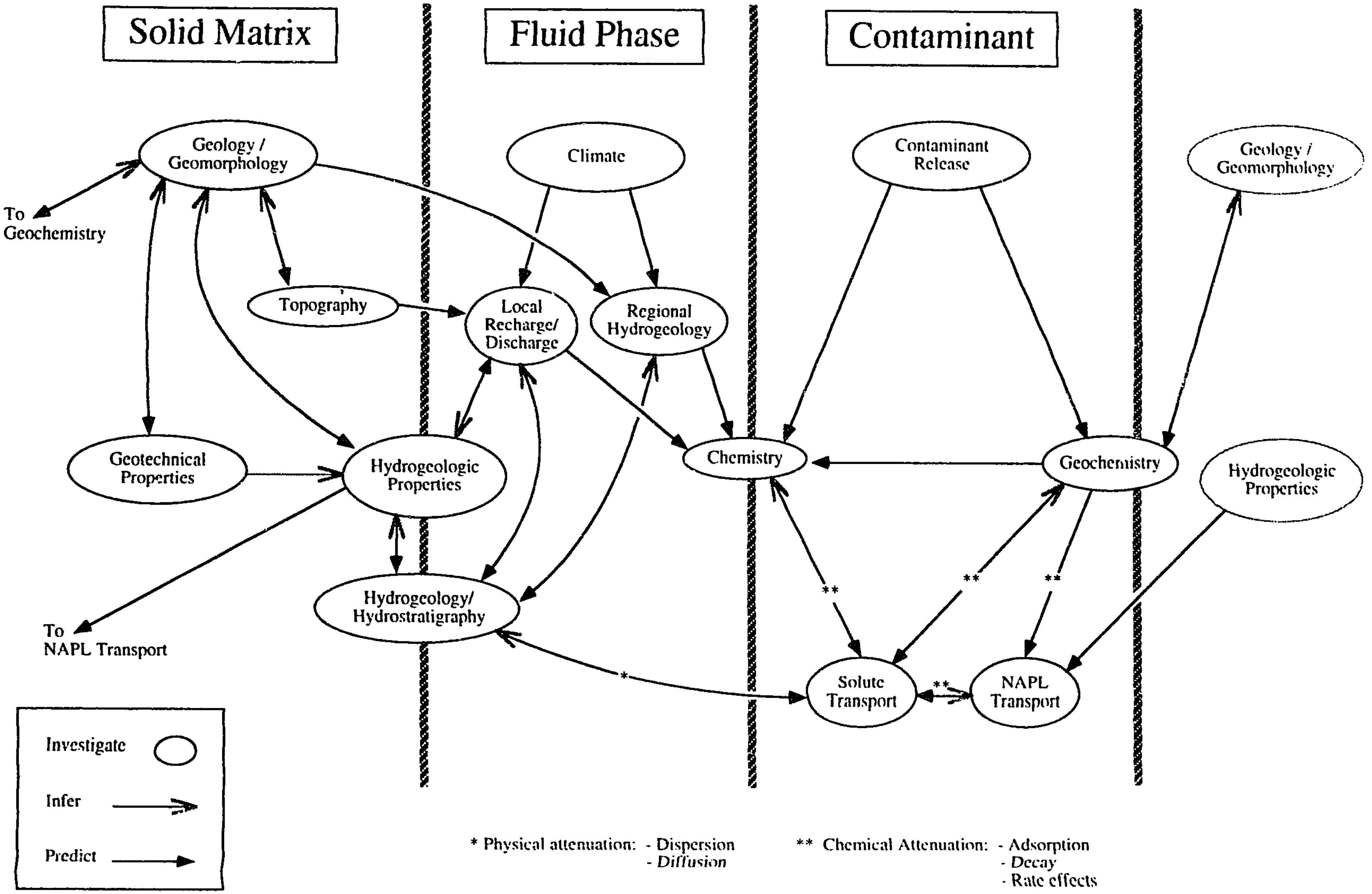
** $K(\theta) = K_0 e^{\beta(\theta)-\theta_0}$

Table 2.2: Typical Length Scales Associated With the Three Fundamental Scales

Case	Dimension of Averaging Volume To Define Point Variables	Variable Characterizing Heterogeneity and Its Correlation Scale I	Usual Dimension of Averaging Volume D	Extent of Flow Domain L
Laboratory	>> molecular	pore scale $I_h = 10^{-3}$	10^{-1}	10^{0}
Local	$10^{-1} - 10^{0}$	log conductivity $I_Y = 10^{0}$	$10^{0} - 10^{1}$	$10^{1} - 10^{2}$
Regional	$10^{1} - 10^{2}$	log transmissivity $I_Y = 10^{3} - 10^{4}$	$10^{2} - 10^{4}$	$10^{4} - 10^{5}$

(after Dagan, 1986) *Length is in me*

Figure 2.1: Flowchart of Property and Process Interrelationships for Contaminant Hydrogeology



* Physical attenuation: - Dispersion
- *Diffusion*

** Chemical Attenuation: - Adsorption
- *Decay*
- Rate effects

CHAPTER 3: PUMP-AND-TREAT CASE STUDIES

3.1 INTRODUCTION

Eleven case studies of pump-and-treat remediation projects have been reviewed and are presented in this chapter. These reports were found in available published literature. Length limitations in many publishing media (journal articles, conference proceedings, etc.) dictated that the information available for site description, contamination delineation, monitoring data, and response to pumping (both hydrologic and contamination extent) was limited. Other sources were found to increase the available detail in some cases. This sometimes involved studies of the contaminant plume separate from the remediation case (eg. Roux and Althoff, 1980), or simply increasing site geology information from general geological sources or specific studies (eg. Owens and Sohl, 1969).

An attempt has been made to characterize the success or failure of these case studies with respect to some single condition or combination of identifiable conditions. This characterization attempt centered on geologic aspects or contaminant properties of the sites, but attempted to encompass all available data.

3.2 SOUTH BRUNSWICK TOWNSHIP SITE, SOUTH BRUNSWICK TOWNSHIP, NEW JERSEY

One of the more extensively detailed case histories of pump-and-treat remediation found in the published literature was that for the South Brunswick Township site. A detailed investigation of the hydrogeology and contamination extent at the site was presented by Roux and Althoff (1980). The remediation system's implementation and the aquifers' initial responses were subsequently described by Althoff *et al.* (1981). This case was of special interest since it illustrated the practice of remediation at the beginning of the 1980's.

3.2.1 General Description and Background

A municipal supply well (SB11) was found to be contaminated with synthetic organic chemicals in December 1977. It was discovered through subsequent investigation that several industrial plants were responsible (in varying degrees) for extensive upper aquifer and limited lower aquifer contamination. This study described the investigation of contamination and the subsequent remedial efforts undertaken for one of these plants (plant B) over the course of time between December 1977 and January 1981.

3.2.2 Site Geology and Hydrogeology

Initial investigation into the extent of the contamination problem proceeded for over a year. The study area was originally much broader than the specific site for which remediation action will be outlined in this study (site B by Roux and Althoff (1980) terminology). Two other major sources of aquifer contamination were identified in that initial investigation (sites A and C) but were not considered here.

Existing lithologic logs from previous well installations (approximately 30 wells) were reviewed. Surveys were subsequently undertaken to redefine the well elevations and water-table distribution. Water samples were taken for analysis with respect to specific organic compounds. Thirty additional investigation wells were then installed around the study area during the investigation. Hollow stem auger drilling was used for the geologic investigation and the installation of these further wells.

Approximately 120 feet (37 m) of unconsolidated sediments overlaid the bedrock in the study area. This area was considered part of the Inner Lowlands subprovince of the Coastal Plain physiographic province (Owens and Sohl, 1969). The stratigraphic sequence determined by Roux and Althoff (1980) during investigation was as follows (descending): Pensauken formation, Old Bridge sand, Woodbridge clay, Farrington sand, and Brunswick formation. The Pensauken formation and the Old Bridge aquifer were undifferentiated sand deposits and have been labelled the upper aquifer for this review. Together, they had a saturated thickness of approximately 35 feet (10.7 m) at the site and consisted of sand, silty sand, and gravel deposits in varying amounts. The Woodbridge clay aquitard was found during investigation to be intermittent in the study area with at least two large openings (inferred from available data) connecting the upper and lower aquifers. This aquitard's mean thickness in the study area varied widely, between 10 and 35 feet (3 to 10.7 m), and occasionally reduced to zero. The Farrington sand was referred to as the lower aquifer. It had varying depth over the Brunswick formation and consisted mainly of sand with some gravel and intermittent clay lenses and streaks. The Brunswick formation, a fractured shale aquifer, was not considered in these contamination investigations.

A further search for available geologic information relating to the site revealed more detailed information on the Pensauken formation and some extra detail on the Woodbury clay. Evidence suggested that the Pensauken formation consisted of fluvial (alluvial or pluvial) deposits laid down in warm interglacial periods. The bedding of the formation was of highly variable thickness and commonly consisted of coalescing lenses. Channel cut and fill actions were postulated as the depositional mode of this facies (Bowman II and Lodding, 1969).

The Woodbury clays were described briefly by Bascom *et al.* (1909) as somewhat micaceous and black. The lower reaches of the deposit were not sandy, with increasing sand in the upper laminated zones. It was noted to be light brown where weathering had occurred. Further, the deposit contained abundant fauna. Owens and Sohl (1969) also noted that it was massive and likely of marine origin.

The hydrogeology of the site was presented in the paper as hydraulic head contours, in plan (Figure 3, Roux and Althoff, 1980). The area surrounding Plant B showed ground water divides to the northeast and east, and elliptical 'windows' in the aquitard below suggested by the 80' head contours in the center of the figure and below the plant. It was mentioned in the paper that the water-table followed the topography of the underlying Woodbury clay to some degree. The hydraulic properties of the upper and lower aquifers are summarized in Table 3.1.

3.2.3 Contaminant Description and Delineation

The contaminants of concern that were detected at public supply well SB11 were synthetic organic chemicals. Primary constituents were 1,1,1-trichloroethane (1,1,1-TCA) and tetrachloroethene (PCE). Trace amounts of benzene, toluene and trichloroethene (TCE) were also detected. The 1,1,1-TCA contamination was found to be the most extensive and was therefore presented in delineation as representing the extent of contamination. This study therefore focused on the 1,1,1-TCA but PCE and trace chemicals were also discussed.

Table 3.2 lists the properties of organic compounds found at the site. It shows that 1,1,1-TCA was ten times more soluble than PCE, with less than half the adsorption affinity to organic carbon (K_{oc}). This compound was also much more susceptible to volatilization and vapour-phase transport. It might be noted that the 1,1,1-TCA would flow through the porous media more easily in free-phase, as suggested by its lower viscosity.

Sampling was done of the decontamination wells, air stripper effluent, and infiltration basins. Monitoring wells were also sampled and tested for the contaminants of concern. Monitoring and delineation of the 1,1,1-TCA plume was done through mixing and sampling the entire saturated depth of the well. Therefore, the concentration readings given in the study represented the minimum possible concentration for the plume at specific horizontal coordinates and times. If the plume's vertical dimension was less than the saturated depth (which was likely) then the actual concentrations were probably higher than the depth averaged samples.

The plume delineation for 1,1,1-TCA previous to remediation system implementation for both sites A and B were presented in the report (Figure 4, Althoff *et al.*, 1981). The 1 ug/l isoconcentration line, taken as the extent of contamination, extended over an area of approximately 640,000 m^2 for plant B. Both plumes shown were directed toward the proposed 'window' in the Woodbury clay aquitard near well SB11. However, the overlaid hydraulic head contours for the upper aquifer did not match the transport pattern of the 1,1,1-TCA plume from plant B. This was apparently due to the change in flow regime caused by SB11's reduced pumping rate after discovery of contamination. The effect of this regime change will be discussed in more detail in Section 4.3.

Most of the domestic supply wells were screened at 60 to 70 feet (18 to 21 m) depths in the upper aquifer. This left them especially susceptible to possible surface contamination in this high permeability aquifer.

3.2.4 Monitoring and Remediation Scheme

Interim action for the site consisted of reduced pumping from the contaminated municipal supply well (SB11) to slow further migration into the lower aquifer. The low level contamination in this effluent was discharged into a municipal sewer system. Possible sources of the contamination, such as sludge ponds and leaking tanks were removed or decommissioned.

Due to use of a similar system at a nearby site, pumping extraction to two in-series air stripping towers was implemented as the chosen remediation system. Seven wells were installed to extract the contaminated water from the vicinity of the original source area. These extraction wells were simply converted monitoring wells that were, as indicated previously, screened over the entire saturated aquifer thickness. A recharge basin was subsequently used to discharge the treated effluent. The total effluent discharge rate was designed to be 198,000 gpd (8.7e-3 m^3/s), though this varied somewhat over the course of the project.

3.2.5 Response To Remediation

Containment of the groundwater contamination was the primary goal of the pumping scheme. Available data was used to model the aquifer response and subsequently estimate the required pumping rate for capture. Approximately 30 % overdesign was included in the estimated extraction rate as a factor of safety. A twenty day test confirmed the effectiveness of drawdown extent in capturing the plume.

After 7 to 10 months of operation the plume size for 1,1,1-TCA was significantly reduced, as defined by the 1 ug/l isoconcentration lines presented in the report (see Figures 1 & 4, Althoff *et al.*, 1981). Based on these figures the areal extent of the plume, defined by the 1 ug/l isoconcentration line, was reduced by roughly 70 % in 7 to 10 months. The absolute concentration values for the plume were reduced to half or less of their initial levels over this same period.

3.2.6 Discussion

The remediation problem addressed in these case studies of 1980 and 1981 reflected the state of the industry during this period. Investigation of the hydrogeologic properties of the subsurface was initially done, focusing on physical transport parameters only. Following this characterization, pumping wells were set up at the source of contamination and downstream to capture and extract the contaminant plume. The contamination levels were monitored in full depth screened wells using depth-averaged sampling techniques. After 10 months of operation the response was extrapolated to predict attainment of clean-up goals in three years (or more).

Subsequent experiences have shown that asymptotic response is typical of pump and treat remediation systems (Hockman, 1992). Later information on the continuing remediation project would have been useful in determining if the three year closure projection of these papers was overly optimistic.

The primary contaminants of concern were the DNAPL's 1,1,1-TCA and PCE. Given the channel-cut-and-fill depositional environment of the aquifer, and the subsequent heterogeneity, withdrawal of residual immiscible fluid contamination in the source area may have been incomplete. Though this would not be apparent in the initial stages of remediation, it would surface as an extensive tailing off period in contaminant withdrawal.

Also, the irregular surface of the Woodbury clay may have given rise to trapped pools of DNAPL that would act as even more persistent sources of continuing contamination.

3.3 CENTRAL NEW JERSEY SITE

This study relied mainly on the information supplied in Herkert and Aslam (1990). Further sources were used as references indicate.

3.3.1 General Description and Background

An industrial site in central New Jersey in operation since the 1960's had contaminated the groundwater in a shallow aquifer with trichloroethene and its degradation products (identified later). The highly heterogeneous nature of the aquifer had led to a problematic remediation operation. A deep aquifer that was heavily utilized (Wenonah-Mount Laurel aquifer) for supply may have been in danger of contamination in the long-term.

3.3.2 Site Geology and Hydrogeology

The site was located in the Atlantic Coastal Plain physiographic province, Central Uplands subprovince. The Upper Cretaceous and Tertiary sediments of this area were of fluvial and marine origin. The sequence of the site was (descending): Vincentown Formation, Hornerstown Formation, and Tinton-Red Bank Formation. The Vincentown Formation was roughly 4 m thick at the site and consisted of largely fine to coarse sands containing varying amounts of silt, gravel and clay. The Hornerstown Formation was roughly 7.5 m thick and contained clayey sands and silts which acted as a confining layer to the Vincentown sands. The Tinton-Red Bank formation consisted of very tight silty clayey sands.

Owens and Sohl (1969) described the Tinton-Red Bank sands and the Hornerstown Formation sands as mainly glauconitic, indicating far shore deposited marine sands. The Vincentown sands were described as mainly quartz sands at the surface, which indicates near shore marine or fluvial deposits. Herkert and Aslam (1990) began investigation of the site in March 1987 with twelve shallow wells installed in the Vincentown Formation and three deep borings. The sharp erosional contact between the Vincentown and Hornerstown Formations found in these borings supported the possible difference in depositional environment (fluvial versus marine) suggested by the mineralogy discussed above.

The site had a shallow ground water-table (approximately 5 feet (1.5 m) depth) and a saturated thickness of only 10 feet (3 m) in the unconfined aquifer. Ground water flow in the area was largely toward the southwest, though the nearby stream did capture some of the adjacent flow. The hydraulic gradient in the water-table aquifer at the site was measured at 0.005 m/m. Porosity was assumed to be 30% for the unconfined aquifer.

The horizontal hydraulic conductivity of the site, measured through a pumping test in 1987, was calculated to be 5.2 ft/day (1.8e-5 m/s). Subsequent investigation, preceding proposed modifications to the remediation project, gave the results shown in Table 3.3. The variation in values based on the method of measurement and location indicated the probable heterogeneous nature of the aquifer. Various monitored well responses to pumping confirmed this. The vertical hydraulic conductivity of the confining layer below (Hornerstown formation, K_v = 0.23 ft/day (8.1e-7 m/s)) was found to be significantly lower than the horizontal conductivity of the aquifer above. Average ground water velocity calculated from the 24 hour pump test conductivity and the average hydraulic gradient was 3.2e-6 m/s.

High heterogeneity was indicated by the changes in conductivity estimates with test method and aquifer volume sampled, though this was not done until unsuccessful containment was apparent. Also, lack of response to pumping from some monitoring wells during tests supported the high heterogeneity hypothesis. Little specific data was given to quantify the variability of hydraulic conductivity in the water-table aquifer. From Table 3.3, calculations of the variance (in log-normal distribution) on a method specific basis gave the following values for the slug test, residual drawdown versus time test, and 24-hour pump test respectively: 1.1, 0.40, and 0.06. These numbers obviously engendered limited confidence given the small number of data points used for each. However, a significant trend might be noted in the heterogeneous variance for the small volume slug test, moving toward a homogeneous variance in the large volume pump test. Though the large volume tests were likely no longer independent.

3.3.3 Contaminant Description and Delineation

The contaminant plume was the result of an accidental trichloroethene spill. No estimate of the total mass was given in the case history and since the soil investigation was unable to locate subsurface source areas, such as residual soil contamination or free-phase NAPL pools, mass estimates were not possible. The contaminants identified in the paper were trichloroethene (TCE) and its various degradation products: 1,1-dichloroethene (1,1-DCE), *cis*-1,2-dichloroethene (*cis*-1,2-DCE), *trans*-1,2-dichloroethene (*trans*-1,2-DCE), and finally vinyl chloride. The last compound was considered the most volatile, mobile, and toxic of the degradation chain. Table 3.4 summarizes some of the characteristics of these compounds that relate to contaminant transport and remediation.

Plume delineation was performed for total volatile organic compounds (VOC's) to account for TCE and all of its degradation products. An initial figure using data from March 1987 (Figure 3, Herkert and Aslam, 1990) showed a plume, delineated by the 100 ug/l isoconcentration line, of approximately 90 x 75 m (6700 m^2). The 1 ug/l line was not shown, but an areal extent of 102,500 ft^2 (9,500 m^2) was given in the text of the report for this concentration.

3.3.4 Monitoring and Remediation Scheme

In March 1988, twenty-one well points were installed to contain and remediate the plume. The capture zone (illustrated in Figure 4, Herkert and Aslam, 1990) would contain the initial plume delineation of March 1987, but not the more extensive plume observed later (April 1989). Eighteen wells were sampled on a quarterly basis to monitor the response of the aquifer to remediation.

The 21 shallow well points installed for interim containment and remediation were also continued as the primary extraction system. These wells and their extraction rate (approximately 11 gpm (6.9e-4 m^3/s) total) were selected without extensive testing of the aquifer hydraulic properties. Later discovery that TCE was migrating beyond the capture zone led to the more detailed investigation described previously and subsequent modelling. Following the various aquifer tests (results shown in Table 3.3) and modelling of the new capture zone, twelve additional pumping wells at the downstream boundary of the site were proposed to attain full containment of the plume. This enhancement had not been implemented at the time of the paper.

3.3.5 Response To Remediation

Drawdown in the water-table aquifer after over two years of pumping operation was shown in the report (Figure 4, Herkert and Aslam, 1990). The capture zone encompassed all of the initial plume extent (Figure 3, ibid.). However, the plume defined by the 1 ug/l isoconcentration line (approximately 9,500 m^2) did not respond as expected. With expanded monitoring wells, a figure from June 1990 (Figure 6, ibid.) showed the concentrations after 27 months of system operation. The appropriate isoconcentration lines were much more extensive than indicated by the previous figure and estimate. An increase in areal extent of approximately 260 % was estimated over the time period between March 1987 and June 1990, based on available figures and data.

3.3.6 Discussion

This study showed limited or even negligible effectiveness of remediation efforts over the report period. The relatively short period of contaminant residence in the subsurface should have been helpful in contaminant removal. However, the heterogeneous nature of the aquifer and the mobility of the free-phase DNAPL probably offset this positive aspect. It should be noted that if DNAPL pools did exist in the subsurface (which seemed likely) the longer they maintained contact with the marine deposited aquitard below, the more likely extensive intrusion and adsorption of DNAPL onto the low permeability soil was to have occurred. This would only further limit success of any complete remediation efforts.

This case appeared to be a classic example of the difficulties in remediation of DNAPL's in heterogeneous aquifers. Source removal was probably inadequate due to an inability to locate possible source masses. The flow channelling observed in various pump test responses had also likely compounded the problematic extraction process. Further details of response to the expanded remediation system would have been useful in interpreting this case more fully.

3.4 FIRESTONE SITE, SALINAS, CALIFORNIA

This case study relied on information supplied by Smedes *et al.* (1993), with other sources as references indicate.

3.4.1 General Description and Background

The study referenced above detailed the closure of hazardous waste areas of a tire manufacturing plant in the Salinas Valley, California. Removal of hazardous substances and contaminated soil, as well as capping of contaminated areas, was involved. It was then discovered that extensive contamination existed in the multi-leveled aquifer system beneath the site and extended beyond site boundaries. The remediation of these aquifers was the focus of the published paper.

3.4.2 Site Geology and Hydrogeology

A survey of available data from existing wells in a 6.4 kilometer radius of the site was initially done. This encompassed 350 logs of varying quality and detail. With the stratigraphic conceptualization derived from this data, four deep stratigraphic wells were installed at key points.

The site was located in the Salinas Valley, considered part of the California Coastal Range physiographic province. The depositional environment of the valley appeared to be meandering stream system alluvial and overbank deposits of sand, silt and clay, locally changing to braided stream deposits of blanket sand and gravel, and occasional estuary deposits of massive clay. Some windblown sand deposits were also in evidence. Also, alluvial fan deposits occurred at the valley margins.

The depositional environment described above had resulted in a system of three aquifers with varying degrees of interconnectivity. Generally the uppermost aquifer, termed the shallow aquifer, was a water-table aquifer confined underneath by estuarine blue clay. This clay layer became intermittent downgradient of the site. The middle aquifer, termed the intermediate aquifer, was also bounded below by a clay aquitard which became discontinuous downgradient of the site. The lowest aquifer system, termed the deep aquifer, was actually a system of several aquifers that had some degree of interconnection in the area of the site. These aquitards all seemed to be commonly continuous below the site and intermittent downgradient.

Generally, the aquifers were composed of sand-gravel channel deposits within a matrix of low permeability overbank deposits. As described previously, these deposits were often meandering stream system or braided stream system deposits, while the aquitards were commonly fluvial clay or estuarine blue clay deposits.

The transmissivities and hydraulic conductivities of the various aquifers were determined through existing pumping data, aquifer tests and other tests of the aquifer materials (i.e. geotechnical tests). The ranges of values found have been presented in Table 3.5.

As described above, the aquitards were considered to be discontinuous at various locations. This was determined through boring, well log investigation and through the similar hydraulic and piezometric head values between the units. The very low permeability of the clay units suggested that contaminant migration was likely to have occurred through the discontinuities rather than directly through the aquitard. Geophysical surveys such as terrain conductivity, resistivity, vertical-gradient magnetometer, natural-gamma-ray, and electric were performed to further delineate the aquifer and aquitard locations and their interconnectivity.

3.4.3 Contaminant Description and Delineation

Initial investigations were undertaken to identify the specific contaminants at this site. The principal ones found were: dichloroethene (1,1-DCE), dichloroethane (1,2-DCA and 1,1-DCA), tetrachloroethene (PCE), trichloroethane (1,1,1-TCA), trichloroethene (TCE), benzene, toluene, ethylbenzene, and xylenes (BTEX). Their properties relating to ground water contamination are shown in Table 3.6.

An initial soil-gas survey was performed in the vadose zone in order to plan the location of sampling and monitoring for contamination delineation. The geophysical surveys mentioned earlier also aided in locating screen depths for monitoring wells.

Sampling of 208 existing wells to delineate the plume laterally and vertically was done. The total number of newly installed wells for site characterization and monitoring was: 189 monitoring wells, 25 extraction wells, and four deep stratigraphic wells. These were installed in the three ground water zones (shallow, intermediate and deep aquifers). The contaminant plume identified for 1,1-DCE was used as the indicator of clean up progress, since it was the last to be removed. The paper showed a schematic of the plume layout and orientation in the three levels of aquifer (Figure 10, Smedes *et al.*, 1993). Approximate sizes of the (elliptical) plumes in the three aquifers was: 1000 by 300 m in the shallow aquifer, 1200 by 300 m in the intermediate aquifer, and 2100 by 450 m in the deep aquifer system.

The paper also showed a schematic of the cross sectional layout of the contaminant plume through the three aquifers (Figure 11, Smedes *et al.*, 1993). The significant downward migration through the aquitard discontinuities indicated by this figure was probably due to the enhanced downward gradient during summer irrigation pumping periods. Gravity driven migration may also have been a factor, though this was not confirmed in the paper. Also, downward migration of source pools should have been considered. This will be expanded upon in the discussion at the end of this case study.

3.4.4 Monitoring and Remediation Scheme

Interim action for the site included source and contaminated soil removal, which occurred before the time period of the paper. Approximately 34,000 m^3 of hazardous liquids and 50,000 m^3 of contaminated soil were removed from the site. The previously contaminated areas were then capped with uncontaminated soil. A soil-gas survey to roughly delineate the contamination extent for investigation was also included as interim action. Six soil-vapour extraction wells were installed in order to control vapour-phase migration and remove contaminants from the vadose zone. At the initial stage, 57 monitoring wells were installed to define the extent of the plume in the shallow aquifer. Finally, fifteen extraction wells were installed on the downgradient edge of the property to contain off-site migration and begin removal of contaminant mass.

As mentioned previously, a total of 189 monitoring wells were installed in the three aquifers for this remediation project. Also, 208 existing wells were initially sampled to delineate the contamination extent. The 25 installed extraction wells were also available for chemical sampling.

In addition to the 15 on-site extraction wells installed in the shallow aquifer for interim containment and remediation, 5 off-site wells were subsequently installed in that same aquifer and 5 wells were installed in the intermediate aquifer. The location of these additional wells was determined through modelling investigation, which utilized the initially available data as well as response data from the interim actions. The total pumping rates used in the model and implementation were dictated by the limitations of the treatment system (650 gpm or 4.2e-2 m^3/s). This treatment system used an air stripper in parallel with an activated-charcoal absorption unit. The effluent from treatment was discharged into the Salinas River bed, which ran dry in the summer time.

3.4.5 Response To Remediation

The initial fifteen extraction wells that were installed across the downgradient edge of the plant property were designed specifically to capture all flow in the shallow aquifer without gaps in coverage. No information was presented as to whether this goal was achieved. However, the amount of monitoring applied to this project could have been expected to detect any inadequate containment.

The report illustrated its stated claim that a 98 % areal reduction of the 1,1-DCE plume was observed over five years of remedial system operation (Figure 7, Smedes *et al.*, 1993). The paper also claimed that the final areal extent of this plume was zero and that cleanup had been achieved. This represented a reduction of all contaminant concentrations to below the set cleanup goals. It was stated that 1,1-DCE was the last compound to attain these goals.

The total mass of contaminants retrieved by the remediation system was approximately 225 kg. Large quantities were speculated to have harmlessly volatilized into the atmosphere through the vapour extraction system, air stripper action, and natural vapour diffusion of the vadose zone.

3.4.6 Discussion

This case study provided an exceptional example of complete remediation goal attainment to the regulatory standards. From the information provided, all contaminants both dissolved and free-phase were successfully removed from the subsurface or diluted to below cleanup goals. Significant dilution and volatilization effects must have been assumed to be at work since the total pore volumes extracted were estimated to be approximately half of the given initial volume of ground water contamination (see Table 3.13).

The possibility of DNAPL source pool migration downward mentioned earlier does not agree with the apparent cleanup goal attainment after only 0.5 pore volumes extracted. Given the number of monitoring wells and the detailed understanding of the subsurface which was apparent in the study, it must be assumed that concentration levels were indeed reduced to acceptable values without subsequent rebound. The scale of the remedial operation to remove less than 230 kg of contaminants (though more may have been volatized) gives this case both a positive aspect (in terms of success) and a negative aspect (in terms of cost efficiency).

An iterative approach to site characterization through progressive improvement in model input was illustrated by this study. The resulting refinement in the site conceptualization allowed the remediation project to be modified and in some cases reduced without loss to the effectiveness and efficiency of the system.

3.5 WOODLAND PARK SITE, WOODLAND PARK, MICHIGAN

This case study relied mainly on information supplied by Posthuma *et al.* (1983). Supplemental sources were used as references indicate.

3.5.1 General Description and Background

A rail accident in February 1978 released vinylidene chloride (1,1-dichloroethene or 1,1-DCE), phenol, and ethylene oxide into the environment. Due to the low environmental temperatures at the time some of the phenol was retrieved in a frozen state on the surface. Most of the ethylene oxide volatilized from the surface and vadose zone and did not apparently create a contamination problem. Soil and groundwater contamination from 1,1-DCE and phenol was discovered through subsequent investigation.

3.5.2 Site Geology and Hydrogeology

Over 200 new borings were performed at the site to characterize the subsurface and monitoring wells were installed to delineate the contamination. The wells were installed in multi-level clusters to track the ground water plumes vertically as well as horizontally.

The water-table aquifer was identified as a glacial outwash deposit of sand and silty sand of approximately 20 m thickness. A few silt, clay, and sand aquitards existed under the spill site at roughly 15 to 20 m depths below the water-table. The uppermost few of these units were discontinuous and disappeared rapidly downgradient of the site. The water-table aquifer was bounded at its lower extreme by the first continuous silt, clay and sand aquitard at 30 m below the water-table. Some silt lenses were also noted within the sand aquifer during investigation. Below the first continuous aquitard was a confined aquifer approximately 30 m thick which was bounded below by an aquiclude of clay and silt.

The general ground water flow was to the south-southwest at a velocity of approximately 1.3 ft/day (4.6e-6 m/s). No hydraulic conductivities were given. However, if a gradient of 0.005 m/m and a porosity of 30 % were assumed (see typical values in Table A.2, Appendix A), a very rough estimate of conductivity would be 3×10^{-4} m/s. Due to the uncertainty in these estimates, an order of magnitude error in either direction for this mean value might be anticipated, though it has been presented for discussion only. The intermittent aquitards and occasional silt lenses mentioned above could have been expected to add to the variability of flow and heterogeneous distribution of conductivities.

3.5.3 Contaminant Description and Delineation

Of the three contaminants spilled at the site, the ethylene oxide was considered to be a negligible threat since it had largely vapourized before causing significant contamination. Some of the phenol and most of the 1,1-DCE did migrate into the subsurface however. Approximately 135,900 kg of 1,1-DCE, 149,500 kg of phenol, and 56,600 kg of ethylene oxide were initially spilled. These were found to exist in the vadose zone as residual contamination and dissolved-phase contamination in the saturated zone. The properties of these contaminants have been described in Table 3.7.

The 1,1-DCE plume horizontal and vertical extents were illustrated in the report (Figures 1 & 2 respectively, Posthuma *et al.*, 1983). The areal extent of the plume was roughly 58,000 m^2 with a pore volume of approximately 230,000 m^3 (assuming a porosity of 30 %) estimated from the given figures.

3.5.4 Monitoring and Remediation Scheme

The 200 monitoring wells mentioned previously were sampled quarterly for contaminant levels. The clustered layout of wells gave both vertical and horizontal delineation of the contaminant plumes. These wells were also monitored quarterly for hydraulic response to pumping. All of this data was subsequently used as input to a numerical ground water model to improve conceptualization of the site.

The remediation system was in operation by September 1979. Seven extraction wells were stated to be installed on the site (though nine were shown on the site layout), four at the downstream end of the 1,1-DCE plume, one at the extreme downgradient edge, and four within the main plume. These wells were designed to operate at a 4.4e-2 m^3/s total pumping rate. Due to concern over the long-term contamination potential from vadose zone residual sources, flushing was initiated to accelerate cleanup of the area under the spill site. This influx of leachate was expected to be drawn into the extraction system in the saturated zone.

3.5.5 Response To Remediation

The purge wells began operation in May 1979 and continued to November 1981. The report noted that the extraction system was designed to totally contain the contaminant plume. However no detail of the success of this containment was given except the efficiency of mass removal implied by the supplied figure (Figure 3, Posthuma *et al.*, 1983). This chart showed that over 99% of the initial mass estimates of 1,1-DCE and roughly 97 % of the phenol were recovered during the operation of this remediation project. These claims were highly dependent on the extracted mass data from the treatment system (aeration ponds and carbon adsorption units) shown in the figure and the estimate of the initial mass spilled

3.5.6 Discussion

This case study showed a highly successful clean-up of soil and ground water contaminated by 1,1-DCE and phenol. One important difference between this and some other more problematic studies was the relatively short residence time of the compounds in the subsurface. Instead of contamination occurring over decades of plant operation, this site was contaminated by one discrete spill of chemicals in the recent past. The high mobility that facilitated the rapid spread of contamination to the ground water probably also facilitated its quick and easy removal. Significant solubility of the contaminants (particularly the phenol) was probably very helpful in pump-and-treat extraction success.

Having discussed possible reasons for the success of this remediation project, it might be appropriate to mention possible errors in the interpretation of responses. The very high percentage mass recoveries were, as indicated previously, calculated dependent on the accuracy of mass removal data and the initial spill mass. However, quantification of the extracted mass being input to treatment systems such as air strippers and aeration ponds might be questionable, depending on the method of sampling and interpreting these values. Also, reported spill masses may have been inaccurate due to conservative estimates on the part of the owner or simple inaccuracies in record keeping.

Finally, these results were encouraging and must not be dismissed simply on the basis of being 'too good to be true'. Quick response, short residence time, highly soluble contaminants, and optimal site conditions (i.e. possible low organic carbon fraction in the aquifer) may indeed have led to such a successful remediation project. However, a critical examination of the reliability of various input data might be useful and appropriate.

3.6 STRASBOURG-ENTZHEIM INTERNATIONAL AIRPORT SITE, ENTZHEIM, FRANCE

This case relied mainly upon information supplied in Zilliox *et al.* (1992), with other sources used as references indicate.

3.6.1 General Description and Background

Contamination of a water supply well near Entzheim international airport was discovered in June 1971. It was determined that the kerosene found in the ground water was from one or several accidental spills during refueling events at the airport.

3.6.2 Site Geology and Hydrogeology

Five observation wells were installed in April 1972 to delineate the extent of pollution. A more detailed investigation was undertaken following discovery of significant free-phase kerosene in the subsurface. In June/July 1973 forty-one augerholes were drilled, with core samples taken periodically. Based on the observation well results, these drillings concentrated on the unsaturated zone between the surface and the water-table.

The area of the site was in the Upper Rhine rift valley, specifically the Alsatian plain aquifer. The aquifer consisted of Rhine alluvial deposits overlain by Vosgian alluvial deposits (Bruche valley). Its stratigraphy was as follows at the site (descending): 1 to 2 m of topsoil (loess), 7 m of coarse sands and gravels with some small pebbles near the lower end, 1.5 m of silt, sandy clays with clay lenses and sand gravel pebbles, a 5 m thick aquifer of clayey sand, gravels and pebbles over coarse sand, gravels and pebbles (Figure 3, Zilliox *et al.*, 1992).

The Bruche valley alluvial deposits were stated to be 10 to 25 times less permeable than the Rhine alluvial deposits. They were roughly separated by the layer of silt and sandy clays with clay lenses at a depth of approximately 8.5 m. The water-table was at 10 m below grade around the site and fluctuated by 0.5 m throughout the year. Hydraulic conductivity in the aquifer was very high, approximately $2x10^{-3}$ m/s, measured at the extraction well. This corresponded to a transmissivity of 1e-1 m^2/s. A storativity of 0.01 was also given.

The silty layer with clay lenses between the two depositional facies had resulted in a hanging capillary zone throughout the subsurface under the site. This had apparently acted as an impermeable barrier between the free-phase kerosene and the water-table aquifer.

3.6.3 Contaminant Description and Delineation

The only contaminant referred to in the study was kerosene, or TR4 aviation turbo fuel. No data was found for material under this (European) designation. Commercial kerosene based aviation fuel is defined by ASTM as Jet Fuel A or A-1. Using the assumption that commercial aircraft fuel is similar in Europe and North America, this designation was used to obtain properties for the contaminant. Table 3.8 gives some properties of this compound.

Estimates of the contaminant mass discharged into the subsurface suggested 3000 m^3 to 6000 m^3, based on contaminated soil volumes and saturation levels. The report showed the initial extent of contamination to be 400,000 m^3 for distinct presence of hydrocarbons (Figure 4, Zilliox *et al.*, 1992).

3.6.4 Monitoring and Remediation Scheme

Intermittent pumping of free-phase kerosene was done from the meteorological station well, that had originally detected contamination, beginning in 1972. This resulted in the recovery of approximately 10 m^3 of free product over 18 months. Previous investigation had revealed the isolated nature of the contamination due to the hanging capillary zone.

Removal of the kerosene was quite problematic. The intermittent extraction system of the meteorological well was replaced with a two pump system for hydraulic containment and free product withdrawal. In the summer of 1973, operation of this system produced some kerosene extraction, but this dropped off over the period of pumping.

A recovery well utilizing a two pump system was installed in 1974 and began operation in March 1975. Ground water extraction for containment began at 60 m^3/h (1.7e-2 m^3/s) with intermittent free product extraction at 0.20 m^3/h (5.6e-5 m^3/s). Over a three year period of operation this system extracted 102 m^3 of kerosene. Free product extraction efficiency dropped off dramatically in 1977, possibly due to hydrogeologic conditions. Redevelopment of the well in August 1977 was followed by increased ground water extraction at 90 m^3/h (2.5e-2 m^3/s).

The attempts at kerosene recovery were stopped in April 1980 and control of drinking water supply wells was taken up as the continued method of contamination management. The rates of pumping from these supply wells was restricted to ensure that extraction from the contaminated plume did not occur. A pumping rate of 4000 m^3/day (4.6e-2 m^3/s) was determined to be the maximum supply well rate allowable for this condition.

3.6.5 Response To Remediation

The containment zone achieved by the recovery well was illustrated in the report (Figure 12, Zilliox *et al.*, 1992). Extraction and reinjection upstream through two paired wells enabled the containment to be maintained without prohibitive effluent treatment requirements.

The volumes of kerosene extracted over the period of free product pumping were small (114 m^3) in relation to the estimated total volume (3000 m^3 to 6000 m^3). The inefficiencies of the extraction system, combined with containment success, low solubility of the contaminant, and underutilization of the Alsatian plain aquifer, probably dictated that attempts at removal be discontinued.

3.6.6 Discussion

This case illustrated what might be termed very limited success in remediation of a large LNAPL contamination problem. The geology of the site, which protected the water-table to some degree with a hanging capillary layer, may also have led to the observed problems in extraction efficiency. This efficiency was likely reduced due to the weak hydraulic connection between the LNAPL pool and the saturated zone. Two pump systems have been found to be quite successful for LNAPL extraction from the water-table in recent years (Penzo and Boyer, 1986). Vapour extraction system (VES) addition might also have been a likely candidate for this case. Unfortunately, no other reports have been found about this site. As a result, it is not known whether any further action, if any, has been taken.

3.7 SOUTHERN ONTARIO SITE A

This study relied on information supplied by Westerby (1992). Other sources of information were used as references indicate.

3.7.1 General Description and Background

The site was a retail gasoline outlet in a residential area. Leaking product delivery lines were found to be causing contamination of the soil, vapours, and ground water below the site and extending off-site. This study described the investigation and remediation of the subsurface for this gasoline contaminated site.

3.7.2 Site Geology and Hydrogeology

Product loss was discovered through inventory examination rather than evidence of soil or well contamination. No known ground water utilization was in operation in the contaminated area. Site investigation involved several backhoe test pits and drilled boreholes. A total of 31 boreholes were subsequently used to install monitoring wells, while four recovery wells were installed in some of the test pits.

The site geologic characterization was based solely on information supplied in the paper since specific locations were not made available to facilitate further characterization. The stratigraphy consisted of (descending): asphalt or concrete, gravelly sand fill, sandy silt with clay layers, fractured sandy silt till with fine sand layers, stratified silt and sand. This stratigraphy covered the upper 10 m of the subsurface. Since the water-table was located at 5 to 6 m depth at this site and the contaminant was a LNAPL, deeper characterization was not supplied.

The water-table was, as mentioned, at 5 to 6 m depths in the vicinity of the site. This located it within the till layer. Specific hydraulic conductivities were not given in this study, therefore containment and withdrawal observations were totally dependent on supplied data.

3.7.3 Contaminant Description and Delineation

The site was contaminated with retail grade gasoline from service station product supply lines. The various properties of gasoline in general and its primary component compounds benzene, toluene, ethylbenzene, and xylenes (BTEX) have been supplied in Table 3.9. Minor (non-aromatic) components cyclohexane, 1-hexene, n-hexene, and heptane (Fetter, 1993) were neglected. Also, it was not specified whether lead contamination was a concern or not, therefore it will not be discussed further in this review.

The initial volumes of spilled contaminant were estimated from product loss records at 10,000 liters (10 m^3). Free liquid product and vapour concentrations were mapped and presented in the report (Figure 1, Westerby, 1992). It was shown in this figure that free-phase gasoline disappeared rapidly beyond the site boundaries. The mass of contaminated soil required for excavation remediation was estimated to be 30,000 tonnes (approximately 15,000 m^3 soil volume). This was determined to be an impractical option but did supply some hint of the contamination extent. The report figure illustrated an areal extent of roughly 4000 m^2 for free product and vapour contamination. The heavy development of the site and surrounding area limited the amount of investigation and plume delineation that was practical. Position of the free product was found to be controlled not only by the water-table position but also by the previously mentioned sand lenses within the till.

3.7.4 Monitoring and Remediation Scheme

During the site investigation free product was manually removed from the observation and recovery wells. This resulted in approximately 3,500 liters (3.5 m^3) of product recovery. The most highly contaminated soil was excavated and removed, in addition to the hydraulic recovery program that constituted the main aspect of this pump-and-treat remediation plan.

Monitoring wells were used to observe free product thickness and vapour concentrations. No measurement of dissolved-phase concentrations in the extracted or in-situ water was mentioned.

The remediation system was designed with single pump wells for product and ground water withdrawal. This option was chosen due to the relatively low permeability of the contaminated soil volume (fractured sandy silt till). The extraction rate was governed by set drawdown of approximately 3 m rather than by specified pumping rates. Without given hydraulic conductivities, estimates of the extraction rate was not possible.

3.7.5 Response To Remediation

The 3 m drawdown at the extraction wells was desired to maximize containment of the contaminated ground water and aid in product recovery. At the time of the report the remediation project had extracted 1,121,000 liters (1,121 m^3) of ground water, of which 15,100 liters (15.1 m^3) was estimated to be liquid product. It should be noted that this volume was 50 % greater than the estimated initial spill volume and did not include whatever volume was contained in the excavated soil. The monitoring wells no longer showed any significant free product on the water surface and the recovered liquid was estimated to be less than 0.5 % gasoline, cessation of pumping operations was therefore planned. Evaluation of a proposed vapour extraction system installation was in process at the time of the report. This second remediation component was being considered due to continued significant vapour concentration levels in some wells, even without evidence of free product.

3.7.6 Discussion

The remediation program appeared to be a success based on the data supplied in the study. Free product recovery of approximately 150 % of initially estimated volumes was accomplished. The lack of free-phase contaminant in the monitoring wells was encouraging. However, continued vapour concentrations and the lack of dissolved-phase monitoring may be a source of concern. The significant solubility of benzene (and secondarily toluene) might have resulted in a dissolved-phase plume supplying the observed vapour-phase plume at the end of the study. Though this was probably not monitored due to a lack of ground water utilization in the area, its potential as a source of vapour contamination might be a problem. Adsorbed contaminants may also exist in the fine-grained till material. Further information on the results of the VES operation, or the proposed continued monitoring, would be informative.

3.8 SOUTHERN ONTARIO SITE B

This study relied on information supplied by Westerby (1992). Other sources of information were used as references indicate.

3.8.1 General Description and Background

This site was also a retail service station located in a residential area. As with the previous site there was no known ground water utilization in the area of the contamination. The contamination was discovered through product loss, which was discovered in September 1988.

3.8.2 Site Geology and Hydrogeology

Eight boreholes were initially drilled on-site to delineate contamination. Only one of these boreholes (at the property's southeast corner) showed any sign of free product. The investigation program extended off-site and eventually encompassed 91 boreholes, most of them monitored. The existence of a channel of higher permeability than the surrounding outwash deposits was identified over the course of this investigation as overlying the aquitard.

The site was located on glacial outwash deposits near a local topographic high. The stratigraphy of the site was as follows (descending): 14 m fine to coarse sand and gravel, a few meters of silty clay, shale bedrock. The sand and gravel layer had moderate to high hydraulic conductivity with much lower conductivity in the underlying silty clay. The water-table was at a depth of approximately 13 m. The investigation revealed a channel of coarser granular deposits directly over the silty clay aquitard. It was postulated that this channel was the dominant migration pathway for contaminants.

Speculation of the geomorphology of the site was difficult without more detailed information or a more specific site location. However, the brief description given above hints at possible kame and esker geology. Fluvial deposition in glacial or post-glacial periods is probable. The nearest surface water drainage was a small creek approximately 1 kilometer downhill of the service station.

3.8.3 Contaminant Description and Delineation

The site was contaminated with retail grade gasoline from a leaking fill-pipe connected to the product tanks. Contaminant properties were similar to those for Southern Ontario Site A. See Section 3.7.3 for details.

Product records were found to be inadequate for accurate calculation of the volumes of gasoline released into the environment. A very rough initial estimate of 60,000 to 70,000 liters was stated in the report.

The investigation described above revealed that the buried channel did indeed control the migration of the free product and dissolved contaminants. The free product plume extended approximately 150 m from the service station along this pathway, with dissolved contaminants extending 300 m beyond this (in rapidly decreasing concentrations). Vapour concentrations were measured up to 350 m beyond the service station though they were found to be confined to the first meter above the water-table. Vapour and dissolved contaminants also extended laterally from the migration channel to some degree.

3.8.4 Monitoring and Remediation Scheme

After the leading edge of the free product plume was identified, a recovery well was installed 110 m downgradient. A two pump system was used in this well, which began operation at the end of October 1988. Investigations continued beyond this date with monitoring wells being installed into 1990.

Approximately 90 monitoring wells in total were screened across the water-table to delineate free- and dissolved-phase gasoline. Vapour concentrations were also monitored through several shallow wells (3 to 6 m) along the buried channel path. Locations of the monitoring wells were constrained by the right of way and development features of the area overlying the narrow, channelized contaminant plume (Figure 2, Westerby, 1992).

The single recovery well, installed in the interim action period, was the only water extraction well used in this operation. A total of four VES wells were installed and operated to control the vapour concentrations observed in the vadose zone. Additional monitoring wells were installed in December 1989 and January 1990 to ensure vapour contamination was being controlled by this system.

3.8.5 Response To Remediation

The higher permeability channel that apparently controlled the migration of contaminants subsequently aided in their containment and extraction. The hydraulic recovery system extracted approximately 63,000 liters (63 m^3) of liquid product from the subsurface. This was accomplished with the extraction of roughly 18,000,000 liters (18,000 m^3) of ground water. The vapour extraction system was estimated to have removed the equivalent of 4,000 liters (4 m^3) of liquid. The total of these recovered volume estimates is in the initial spill volume's estimated range of 60 to 70 m^3. Unfortunately, lack of reliable estimates of the initial volumes of gasoline released from the site prevent accurate overall efficiencies from being calculated.

The VES was shut down in April 1990, after continued monitoring showed low to non-detectable levels for three consecutive months. Continued monitoring for a year after this indicated no further contamination effects.

3.8.6 Discussion

The hydrogeology and hydrostratigraphy of the subsurface in this study not only confined the contaminant migration to a narrow channel, it also aided in the extraction of free and dissolved product plumes. Monitoring of the extensive distribution of monitoring wells (from the problematic site investigation) was helpful in assuring that no contaminant sources remained. Data from both ground water quality and vapour quality sampling for a year after the remediation project ceased showed no further contamination. This was especially helpful since lack of data on initial spill volumes made efficiency and completeness of the extraction system impossible to estimate. It might be suggested from this response that the intial estimates of released volume were reasonably close, though dilution may have aided the reduction process.

3.9 SAN FRANCISCO BAY AREA SITE, SANTA CLARA VALLEY, CALIFORNIA

This study relied on information supplied in Donovan and Murray (1985). Further information was obtained from sources as indicated.

3.9.1 General Description and Background

An underground solvent storage tank leaked solvent contaminants into the ground and ground water over an estimated period of two years. Removal of the leaking tank and remediation of the contaminated subsurface followed the suspected release period within a few years.

3.9.2 Site Geology and Hydrogeology

The investigation of the site consisted of 12 shallow borings with installed observation wells, and two deep borings. The drilling was done with a hydraulic mud rotary drill and split spoon samples taken periodically.

The site was on the margin of the Santa Clara valley alluvial deposits of sands and silts. Cut and fill operations had redistributed some of the soil material on site. The stratigraphy in the contaminated portion of the site was as follows (descending): 5 to 10 feet (1.5 to 3 m) of fill, 4 to 10 feet (1.2 to 3 m) of alluvial terrace deposits of interbedded silty clays and silty sands (downgradient of spill site), Santa Clara formation of interbedded sands, silts, gravel, and clays, Miocene marine sandy siltstone and silty sandstone bedrock. A 3 to 6 feet (0.9 to 1.8 m) thick confining clay layer in the Santa Clara formation was located in the contaminated area at 5 feet (1.5 m) below the water-table. The water-table depth was approximately 30 feet (9 m) below grade in the Santa Clara formation.

The Santa Clara formation's hydraulic conductivity was measured in-situ through rising head permeability tests. This was a heterogeneous unit with results that ranged from 2.7e-7 to 5.1e-6 m/s in the area of the spill, the highest reading was 1.4e-5 m/s at a well downstream of the contamination (well B-4). This location was eventually used as the extraction well. The confining aquitard vertical conductivity was measured in the laboratory to be 8.4e-11 to 4.7e-10 m/s.

3.9.3 Contaminant Description and Delineation

The contaminants of concern at this site were trichloroethene (TCE) and xylenes. The values of various properties for these compounds are supplied in Table 3.10. It should be noted that though xylene has a relatively low water solubility; it is highly soluble in TCE. This will be discussed later.

The delineation of the solvent plume was done with the limited data available from the 12 shallow observation wells. The area of contamination (roughly 3000 m^2) was indicated in the report (Figure 1A, Donovan and Murray 1985). Vertical distributions of TCE and xylenes before remediation were also shown in that paper (Figure 2 & Table 1, ibid.). Though some contamination was noted below the confining clay layer it became apparent that most was restricted to the water-table aquifer above and within the clay aquitard. The pore volume of contamination was estimated to be roughly 1350 m^3 from the above area, using a saturated thickness estimate for the aquifer (1.5 m), and a porosity of 30 %. If the aquitard were included in this estimate, at a thickness of 1.2 m and a porosity of 40 %, the total pore volume increased to 2800 m^3.

Measured concentrations of TCE and xylenes both exceeded the solubilities listed in Table 3.10. It should be noted that the concentrations given were soil concentrations of contaminant per unit soil mass. These values therefore included dissolved-phase, adsorbed-phase, and residual or free-phase contaminants; and thus solubilities do not directly restrict them. The report also delineated dissolved-phase distributions of TCE and xylenes before and after remediation implementation (Figure 1B, Donovan and Murray, 1985). Significant reductions of each contaminant were indicated at the four observation wells that were measured for both time periods.

The information given in the paper suggested that approximately 1 to 2 US gallons (3.8 to 7.6 liters) per week were released into the ground over a period of about two years. These numbers produce an estimated total release volume of 0.4 to 0.8 m^3.

3.9.4 Monitoring and Remediation Scheme

The source of contamination was a leaking underground solvent storage tank. This tank was excavated in April 1981, approximately two years after product loss began. Soil investigation revealed that concentrations of contaminants in the vadose zone were significantly lower than those below the water-table. This was interpreted to indicate that the compounds had largely migrated into the saturated zone.

Ground water extraction was limited by the site's stratigraphy. The saturated depth above the clay aquitard was only 5 feet (1.5 m). Therefore, the pumping rate at the one extraction well was set by the limiting drawdown of 4 feet (1.2 m). The limited drawdown, combined with relatively low permeability in the saturated zone, resulted in a predicted extraction rate of 0.1 gpm (6.3e-6 m^3/s). The actual extraction rate, measured in well B-4 downstream of contamination source and in a higher permeability zone, was 0.4 to 0.5 gpm (2.5e-5 to 3.2e-5 m^3/s). The relatively low permeability of the aquifer resulted in slow capture zone development. However, it was eventually determined that the single well would capture the entire contaminated plume.

3.9.5 Response To Remediation

As mentioned above, the capture zone was thought to contain the entire contaminated plume. The water-table response to this pumping effort was shown in the paper (Figures 4 & 5, Donovan and Murray, 1985). The delineated capture zone in these figures was developed after six months of continuous pumping.

Donovan and Murray (1985) gave limited data as to withdrawal response of the remediation system. The paper does show measured aqueous concentrations in 1981 and in 1984 after three years of continuous pumping (as previously mentioned). Significant decreases in concentrations were measured in all monitoring wells over this period, though none had reached drinking water standards as of 1984. Areal reduction of the contaminated plume was not supplied in this paper.

3.9.6 Discussion

This study showed positive response of the saturated zone to ground water pumping for containment and extraction. The aqueous concentration decreases were encouraging. However, the heterogeneous nature of the aquifer, and the problematic nature of the LNAPL xylenes and the DNAPL TCE, dictate further observation of continued remediation to assure that cleanup goals were being approached. The estimated release volumes mentioned earlier (0.4 to 0.8 m^3) might be compared to extraction data if this had been made available.

The contact time of the contaminant plume with the aquitard may have resulted in diffusive transport of dissolved contaminants into its volume. If this had occurred in significant masses, the dissolved and adsorbed mass within the clay may have acted as a continued source of contamination for many years. This contamination may not then become apparent until the Santa Clara formation itself was cleaned to a lower level.

3.10 VERONA WELL FIELD SITE, BATTLE CREEK, MICHIGAN

This study relied on information supplied in Caplice and DeFouw (1986). Other sources were used as references indicate.

3.10.1 General Description and Background

The site was surrounded by a rural area to the north and east, light to heavy industrial areas to the south and west, and a residential area directly to the south. Rail yards were located on the eastern edge of the site. Contamination of the well field's ground water with volatile hydrocarbons was discovered in August 1981. Subsequent investigation revealed 10 city supply wells to be contaminated. It was also found that 80 private wells were contaminated with the same compounds.

3.10.2 Site Geology and Hydrogeology

The U.S. EPA began a remedial investigation in November 1983 which initially defined the horizontal and vertical extent of contamination through soil and water sampling. The source of contamination was then determined to be a nearby solvent company.

Limited information was supplied about the subsurface geology. The aquifer in question was identified as a fractured sandstone unit called the Marshall sandstone formation. Given the extensive usage of the aquifer as a ground water supply source, it was assumed that it was of significantly high yield.

3.10.3 Contaminant Description and Delineation

The major contaminants of concern were identified as 1,1,1-trichloroethane (1,1,1-TC trichloroethene (TCE), perchlorethene (PCE), and 1,2-dichloroethane (1,2-DCA). properties of these compounds relating to ground water contamination are liste 3.11.

The total volatile organic compounds (VOC's) found during investigations were in Caplice and DeFouw (1986). Figures 3 & 4 of this paper showed the migr contamination toward the Verona well field. Delineation upgradient of well V35 was not presented, restricting estimation of the contaminated plume area. However, the indicated area did grow by roughly 110 % between January-April 1982 and September 1983-January 1984.

3.10.4 Monitoring and Remediation Scheme

A five well purge wall was installed and activated on May 25, 1984. Treatment of the effluent from this system was intended for an air stripper unit. However, for interim treatment (until the air stripper could be completed), granular activated carbon (GAC) units were installed. These units operated until the air stripper was brought on line September 19, 1984 and continued for one month after that date for debugging of the new system.

One of the potentially responsible parties (PRP's) was ordered to purge a layer of immiscible contaminants below its facilities. This was done from February 1984 until spring of 1985 when bankruptcy of the company resulted in a lapse of the monitoring program. The remediation system at the Verona site consisted only of the purge well barrier mentioned previously.

3.10.5 Response To Remediation

The sole purpose of the remedial operation detailed in Caplice and DeFouw (1986) was to halt the VOC's migrating toward the city supply well field. Water-table contours given for February 1984 and August 1984 in the paper illustrate that hydraulic containment was achieved. Effluent concentrations in three wells beyond the purge barrier were charted in the study to fall off significantly after start up of the system. Since the main purpose of the remedial operation was to contain the contaminant migration, plume responses were not given.

3.10.6 Discussion

This study focused on interim (containment) action of a contamination problem. Also, the perspective of the paper was that of construction management rather than environmental impact. As such, much detail was supplied of construction details and costing, with little given of subsurface conditions or contaminant plume responses. The limited detail in subsurface geology supplied by the paper made speculation of the long-term effects of this contamination extremely difficult. Effectiveness of the responsible party's purging process would have been valuable in determining the potential for continued contaminant mass supply to the aquifer.

The contaminants in this study were largely DNAPLs. Given the fact that the aquifer was a high yield sandstone formation, it might be hypothesized that some pooled contaminant resided either on the aquifer surface or in fracture networks. Remediation of such scenarios have been considered very difficult in the past without other beneficial geologic or hydrogeologic features.

3.11 NEW ENGLAND SERVICE STATION SITE

This study relied on information supplied by Penzo and Boyer (1986). Other sources of information were used as references indicate.

3.11.1 General Description and Background

The subsurface at an independently owned service station was found to be contaminated with petroleum upon a routine real estate transfer site assessment. It was discovered upon subsequent testing that two unleaded gasoline storage tanks were leaking.

3.11.2 Site Geology and Hydrogeology

Seven new monitoring wells were installed at the site. Borehole logs from these installations were used to aid in conceptualizing the subsurface conditions. The wells were screened across the water-table to monitor any free floating LNAPL (gasoline). The site was described as being located over a coarse-grained, high yield glacial aquifer approximately 1/4 mile (400 m) upgradient of a lake. The lake served as a municipal water supply for the small community surrounding the site.

The transmissivity of the aquifer beneath the site was measured through an eight hour pumping test to be 250,000 gpd/ft (4.3e-2 m^2/s). Hydraulic conductivity was subsequently given as 5,000 gpd/ft^2 (2.8e-3 m/s; implying an average saturated thickness of 15.3 m). The natural ground water velocity was calculated to be 2 ft/day (7.1e-6 m/s). It was supplied in the report that the free-phase hydrocarbon flowed at approximately one-tenth the rate of the ground water. This gave it a calculated velocity of 0.2 ft/day (7.1e-7 m/s). Natural ground water flow was west-northwest under the site.

3.11.3 Contaminant Description and Delineation

The site was contaminated with retail grade unleaded gasoline from leaking product tanks. Contaminant properties were assumed similar to those for Southern Ontario Site A. See Section 3.7.3 for further details and Table 3.9 for compound properties.

The extent of the dissolved hydrocarbon plume on November 1, 1985 was presented in Penzo and Boyer (1986). The area within the roughly circular 1000 ppb contour for dissolved hydrocarbons in this figure was calculated to be 1400 m^2.

3.11.4 Monitoring and Remediation Scheme

The source of hydrocarbons into the subsurface was determined to be two leaking gasoline storage tanks. These were removed immediately upon discovery. A continued source of ground water contamination was the free product plume floating on the water-table. Extraction of this mass was accomplished through use of the two pump well system described below.

The dissolved contamination levels in the subsurface were measured through sampling six of the monitoring wells. A single recovery well was located at the center of the initial plume discovered in the investigation. The completion depth of this well was 48 feet (14.6 m). The two pump system utilized in this well consisted of a ground water extraction pump which created a drawdown cone for free-phase containment and dissolved-phase withdrawal, and a free-phase pump at the water surface to extract the NAPL (gasoline).

3.11.5 Response To Remediation

Containment achieved by the drawdown cone of the extraction well was represented by the pumped water-table shown in the paper (Figure 5, Penzo and Boyer, 1986). From this and plume delineation figures, full containment of the plume was not indicated. However, no other containment actions were discussed. The design flow rate of the groundwater extraction pump was 50 to 100 gpm (3.8e-3 to 7.6e-3 m^3/s).

Areal reduction of the contaminated plume was illustrated by the change between two figures in the report (Figures 4 & 6, ibid.). The reduction was from 1400 m^2 to 740 m^2, or roughly 50 % by area. The text of the study claimed 90% reduction in observed dissolved hydrocarbon concentrations over a 30 day period of system operation. Also, 1000 imperial gallons (4.5 m^3) of free-phase gasoline were extracted.

3.11.6 Discussion

This study showed rapid initial success in removing both free-phase and dissolved-phase gasoline. However, this initial positive response is a common scenario for pump-and-treat remediation systems. More critical responses have often been extraction efficiencies in the later stages of clean-up. With the coarse-grained very high permeability aquifer at the site, pumping extraction performance may indeed have held up to the initially experienced response. Additional VES remediation might have been helpful for the volatile compounds in question, though the development of the area may have made this type of system's effluent concerns difficult to resolve.

3.12 COASTAL AIRPORT SITE, ATLANTIC PROVINCES, CANADA

This study relied on information supplied by Penzo and Boyer (1986). Other sources of information were used as references indicate.

3.12.1 General Description and Background

The site consisted of a man-made aquifer contaminated with jet fuel through a line leak. Free-phase hydrocarbon containment and removal using a two pump extraction well was done. However, no mention of dissolved hydrocarbon concentrations was made in the study. The ground water effluent was discharged to a nearby stream.

3.12.2 Site Geology and Hydrogeology

Nine existing monitoring wells were utilized in the initial site investigation. Stratigraphy and geology of the aquifer were well known since it was of anthropogenic origin. It was known to be an aquifer of moderate to high yield sand and gravel. Eight new boreholes/wells were installed as part of the investigation program. Most were drilled to 25 feet (7.6 m), with one deep boring extending to 37 feet (11 m; the expected lower boundary of the aquifer). A clay barrier (aquitard) beneath the aquifer was also anthropogenic. No boreholes penetrated this stratigraphic unit in order to avoid cross contamination of the (natural) underlying aquifer.

A pumping test to determine the hydraulic properties of the contaminated aquifer gave a transmissivity of 100,500 gpd/ft (1.7e-2 m^2/s). This converted to a hydraulic conductivity of 3865 gpd/ft^2 (2.2e-3 m/s; implying an average saturated thickness of 7.9 m). The natural ground water velocity was given as 4.3 ft/day (1.5e-5 m/s), with the hydrocarbon free-phase velocity roughly 1/10 of that value, as in the previous study.

3.12.3 Contaminant Description and Delineation

The site was reported to be contaminated with jet fuel. Properties of a common military jet fuel mixture (JP4) are listed in Table 3.12. In some categories (as noted), JP4 was assumed to have similar properties to ASTM Jet Fuel B due to limited information. Both are wide-cut fuels of kerosene and gasoline mixture. However, limited confidence can be given to these assumed properties and they are therefore presented as the best available estimates for discussion only.

No information was given on the dissolved plume for this study. The dimensions of the free-phase hydrocarbon plume were illustrated in Penzo and Boyer (1986). It was shown to cover roughly 4700 m^2 to the 0.5 feet (0.150 m) thickness contour. The total volume of spilled fuel was given as 100,000 imperial gallons (450 m^3).

3.12.4 Monitoring and Remediation Scheme

The single extraction well installed to remediate this site was predicted to contain the free-phase petroleum plume except that portion extending 56 feet (17 m) downgradient of the well location. No discussion of further containment efforts was given in the study. Also, the only stated contamination concern was that of nearby coastal fishing beds.

As mentioned above, no dissolved-phase monitoring results were given in the study. However, reduction of the free-phase hydrocarbon plume was clearly illustrated. The two pump system of drawdown containment and free-phase extraction used in the previous study was again applied to this one.

3.12.5 Response To Remediation

The paper illustrated the reduced free-phase hydrocarbon plume after 62 days of pumping system operation (Figure 10, Penzo and Boyer 1986). Little areal reduction (to the 0.5 feet contour) of the plume was apparent from this figure. However, it was reported that 14 monitoring wells showed a decrease in plume thickness and the leading edge of the plume was being reversed toward the extraction well. Approximately 16,000 imperial gallons (73 m^3) had been recovered in that time period.

3.12.6 Discussion

The limited time period of the study did show promising extraction results for the system. However, as in the previous case initial results must be followed up with long-term extraction efficiencies. In this case the dissolved-phase plume was not monitored. The limited solubility of this contaminant (see Table 3.12), as well as the lack of utilization of the aquifer and its isolation from the natural aquifer below, perhaps resulted in limited concern over this contamination aspect.

Due to the anthropogenic nature of the aquifer and aquitard below, it was probable that both units were relatively homogeneous by natural formation standards. This may have resulted in optimal results for the remediation project. Further data would be required to confirm this speculation.

3.13 SUMMARY

The effectiveness of the pump-and-treat remediation projects presented in this chapter are summarized in Table 3.13. These studies might be separated into three categories: successful remediation projects, promising initial results (subsequent data needed), and unsuccessful remediation projects.

The successful projects included: the Firestone site, the Woodland Park site, Southern Ontario site A, and Southern Ontario site B. The Firestone site was stated directly in the report as having achieved cleanup objectives. In the case of the catastrophic release at the Woodland Park site, the data supplied indicated approximately 98 % removal of the initially spilled mass of contaminants. Also, the ground water aquifer discharged directly into a marshy area where natural degradation should have alleviated any residual contamination. Both Southern Ontario sites extracted more than the initial estimates of contaminant mass. Site A was reduced to very little free-phase gasoline in the subsurface, and a VES system (with continued monitoring) was planned to insure successful goal attainment. Site B was monitored for one full year after closure of both pump-and-treat and VES components of the project with no evidence of continued contamination.

The sites requiring follow up studies to clarify promising initial results include: South Brunswick Township, San Francisco Bay Area, New England, and Atlantic Coastal Airport. In the South Brunswick Township site the initially promising 70 % areal reduction of the plume was extrapolated at the time of the report (1980-81) to a three year closure schedule. The 85 % average concentration reduction in monitoring wells for the San Francisco Bay Area site also led to optimistic projections. However, both of these sites dealt with problematic DNAPL contamination in soil material with varying degrees of heterogeneity. Any projections based on initial responses must therefore be cautiously made. The New England service station involved volatile LNAPL (gasoline) contaminants. Addition of a VES to this site would likely improve remedial success if any further difficulties were encountered. The Coastal Airport site was contaminated with low volatility LNAPL (aviation jet fuel, JP4). However, the probable homogeneous nature of the anthropogenic aquifer and aquitard might aid in the attainment of cleanup goals for the complete project. The positive initial responses of these last two cases might therefore be taken with more optimism. However, care should be taken in projecting long-term response from promising initial results. The typical asymptotic response observed in most ground water extraction systems, discussed in Hockman (1992) and illustrated in Figure 3.1, supports this caution.

Two projects that might be considered failures were the Central New Jersey site and the Strasbourg-Entzheim Airport site. The Central New Jersey site showed a negative response to (or in spite of) pump-and-treat remedial efforts. This case was one of highly difficult conditions: heterogeneous aquifer material, DNAPL contaminant with probable free-phase pools, and limited characterization of the lower boundary material and its interface with the aquifer. The Strasbourg-Entzheim site showed very inefficient (though slightly positive) response to pump-and-treat efforts. This low response was likely due to the lack of hydraulic connection between the contaminated zone and the pumped aquifer. The remedial effort was subsequently discontinued without goal attainment in favour of well resource management in the area.

The Verona well field site was designed as only a containment project. In this respect, it appeared to have succeeded in attaining its objectives. However, the focus of the study was on construction and management details rather than hydrologic features and responses. This therefore left significant gaps in the data that would be required for interpretation of the project's responses. Further information would be needed to hypothesize any degree of success in this case.

A brief review of the information supplied in Table 3.13 does not enable successful and unsuccessful projects to be readily differentiated or predicted, other than in their shown performances. Most of the cases were coarse-grained aquifers, a common prerequisite in application of pump-and-treat technology. Fluvial deposits experienced both successful and unsuccessful remedial efforts. Also, both DNAPLs and LNAPLs were sometimes successfully and sometimes unsuccessfully extracted.

It appears that the successful attainment of cleanup [illegible] in each case was often dependent on one or two site specific dominating feature, rather than one single identifiable characteristic. In terms of the failures, some were more traditionally understood as problematic, such as DNAPL contamination of a highly heterogeneous aquifer (as in the Central New Jersey site). Other failures were apparently due to more unique conditions, such as the hanging capillary zone of the Strasbourg-Entzheim site which hydraulically isolated the contaminated zone from the saturated zone. In the case of the Woodland Park site, the most easily identifiable feature to which to attribute success would be the short residence time of the contaminant in the subsurface. In the case of Southern Ontario site B, the success might be attributed to the coarser grained channel which dominated contaminant migration in both directions (and in both free and dissolved phases).

It should be noted that the case studies available in literature do not necessarily give an overall view of the state of pump-and-treat remediation applications. Unsuccessful cases would tend not to be published, as this would not usually be in the best interests of the owner or funding source, unless some other factors worthy of discussion were discovered. There is rarely any motivation to attract publicity to the remediation of contaminated sites, in today's social and political climate, unless the results are overwhelmingly positive. Also, the results given in terms of concentration reduction or reduction in plume extent (defined by the 1 ppb isoconcentration line, the detection limit, etc.) do not usually describe the monitoring, sampling, or analysis protocol in enough detail to fully assess the appropriate confidence to be assigned them. These difficulties were discussed in greater detail previously (Section 2.6.3).

Table 3.1: Hydraulic Properties of Upper and Lower Aquifers, South Brunswick Township site

Aquifer	Hydraulic conductivity, K_h	Ground water velocity,	Hydraulic gradient, i	Other
	gpd/ft^2 (m/s)	ft/day (m/s)	ft/ft (m/m)	
Upper aquifer (at site)	1,600 to 3,500 (7.5e-4 to 1.7e-3)	5 (1.8e-5)	0.009 to 0.004	porosity = 0.40
Upper aquifer (beyond site)		1 to 2 (3.5e-6 to 7.0e-6)		
Lower aquifer	2,000 to 3,000 (9.4e-4 to 1.4e-3)	0.3 to 6 (1.1e-6 to 2.1e-5)	0.0004 to 0.005	S_y = 0.32

after Roux and Althoff (1980).

Table 3.2: Contaminant properties for South Brunswick Township site

Contaminants	S_G	σ_{IFT}	S_o	vp	H	log (K_{oc})	V	MCL **
		(dynes /cm)	(mg/l)	(mm Hg)	(atm m^3/mol)	(ml/g)	(m^2/s)	(ug/l)
1,1,1-TCA	1.34	(28.28-0.12*T)	4400	100	2.01e-1	1.78	4.1e-3	200
PCE	1.62	32.86	150	14	2.59e-2	364	1.2e-5	5
Trace contaminants:								
TCE	1.47	29.5	1100	57.9	9.10e-3	126	3.9e-6	5
toluene	0.86	(30.9-0.12*T)	535	28.1	6.37e-3	300	6.4e-6	1000
benzene	0.87	28.9	1750	95.2	5.59e-3	83.0	6.9e-6	5

from Simms et al. (1991) except σ_{IFT} *- Mercer and Cohen (1990) and ** (USEPA from Fetter (1993) and DeZuane (1990)).*

Table 3.3: Hydraulic conductivity values for central New Jersey site

METHOD	MW-2	MW-3	MW-13	MW-14	MW-15	MW-16	MW-17	AVG.
Distance-Drawdown Analysis of Long-term Pumping Data	-	-	-	-	-	-	-	25 (8.8e-5)
Slug Test	-	2.1 (7.4e-6)	1.8 (6.4e-6)	0.23 (8.1e-7)	2.8 (9.9e-6)	2.7 (9.5e-6)	0.68 (2.4e-6)	1.7 (6.0e-6)
Residual Drawdown Versus Time	4.4 (1.6e-5)	3.8 (1.3e-5)	4.7 (1.7e-5)	0.95 (3.4e-6)	-	3.2 (1.1e-5)	1.7 (6.0e-6)	3.1 (1.1e-5)
24-Hour Pump Test	52.4 (1.9e-4)	-	-	62.3 (2.2e-4)	-	37.8 (1.3e-4)	62.3 (2.2e-4)	53.7 (1.9e-4)

adapted from Herkert (1990), in ft/day (m/s).

Table 3.4: Contaminant properties for Central New Jersey site

	S_G	σ_{IFT}	S_o	vp	H	(K_{oc})	V	MCL **
Contaminants		(dynes /cm)	(mg/l)	(mm Hg)	(atm m^3/mol)	(ml/g)	(m^2/s)	(ug/l)
TCE	1.47	29.5	1100	57.9	9.10e-3	126	3.9e-6	5
1,1-DCE	1.22	24	2250	600	3.40e-2	65.0	3.0e-6	7
cis-1,2-DCE	1.27	24	3500	208	7.58e-3	49.0	3.5e-6	70
trans-1,2-DCE	1.25	24	6300	324	6.56e-3	59.0	3.2e-6	100
vinyl chloride								2

*from Simms et al. (1991) and Mercer and Cohen (1990), except ** (USEPA from Fetter (1993) and DeZuane (1990)).*

Table 3.5: Hydraulic properties of the various Salinas Valley geologic units

	Hydraulic Conductivity	Transmissivity
Aquifer / aquitard	ft/day (m/s)	ft^2/day (m^2/s)
Shallow aquifer	100 (3.5e-4)	(7.1e-3 estimated)
Intermediate aquifer	200 to 1,200 (7.1e-4 to 4.2e-3)	10,000 to 50,000 (1.1e-2 to 5.4e-2)
Deep aquifer system	200 to 1,200 (7.1e-4 to 4.2e-3)	
Various clay aquitards	3e-4 to 8e-6 (1.1e-9 to 2.8e-11)	

Table 3.6: Contaminant properties for the Salinas Valley site

Contaminants	S_G	σ_{IFT}	S_o	vp	H	(K_{oc})	V	MCL **
		(dynes /cm)	(mg/l)	(mm Hg)	(atm m^3/mol)	(ml/g)	(m^2/s)	(ug/l)
1,1-DCE	1.22	24	2250	600	3.40e-2	65.0	3.0e-6	7
1,2-DCA	1.26	(35.43-0.14*T)	8520	64.0	9.78e-4	14.0	7.0e-6	5
1,1-DCA	1.18	(27.03-0.12*T)	5500	182	4.31e-3	30.0	4.3e-6	-
PCE	1.63	32.86	150	17.8	2.59e-2	364	1.2e-5	5
1,1,1-TCA	1.35	(28.28-0.12*T)	1500	123	1.44e-2	152	6.7e-6	200
TCE	1.47	29.5	1100	57.9	9.10e-3	126	3.9e-6	5
benzene	0.87	28.9	1750	95.2	5.59e-3	83.0	6.9e-6	5
toluene	0.86	(30.9-0.12*T)	535	28.1	6.37e-3	300	6.4e-6	1000
ethylbenzene	0.87	(31.48-0.11*T)	152	7.00	6.43e-3	1100	7.8e-6	700
xylenes †	0.87	(31.48-0.11*T)	168	8.9	7.6e-3	894	7.9e-6	10,000

*from Simms et al. (1991) and Mercer and Cohen (1990), except ** (USEPA from Fetter (1993) and DeZuane (1990)).*

† *average of m-xylene, o-xylene, p-xylene values.*

Table 3.7: Contaminant properties for the Woodland Park site

Contaminants	S_G	σ_{IFT}	S_o	vp	H	(K_{oc})	V	MCL **
		(dynes /cm)	(mg/l)	(mm Hg)	(atm m^3/mol)	(ml/g)	(m^2/s)	(ug/l)
1,1-DCE	1.22	24	2250	600	3.40e-2	65.0	3.0e-6	7
phenol †	1.24	(42.5-0.11*T)	290,000	1.80	1.05e-5	400	1.8e-5	††

*from Simms et al. (1991) and Mercer and Cohen (1990), except ** (USEPA from Fetter (1993) and DeZuane (1990)).*

† *2-chlorophenol values used.*

†† *No drinking water standards found, though pentachlorophenol has a secondary MCL of 30 ug/l; WHO (1984) suggests total phenol concentration below 1 ug/l.*

Table 3.8: Contaminant properties for Strasbourg-Entzheim site

Contaminants	S_G	σ_{IFT}	S_o	vp	H	(K_{oc})	ν	MCL **
		(dynes /cm)	(mg/l)	(mm Hg)	(atm m^3/mol)	(ml/g)	(m^2/s)	(ug/l)
kerosene (aviation turbo fuel TR4)	0.78 to 0.84	-	30	2.5	-	-	1.5e-6	-

from Kirk-Othmer (1978).

Table 3.9: Contaminant properties for Southern Ontario sites A & B

Contaminants	S_G	σ_{IFT}	S	vp	H	K_{oc}		MCL **
		(dynes /cm)	(mg/l)	(mm Hg)	(atm m^3/mol)	(ml/g)	(m^2/s)	(ug/l)
gasoline (automobile)	0.73	21						
benzene	0.87	28.9	1750	95.2	5.59e-3	83.0	6.9e-6	5
toluene	0.86	(30.90-0.12*T)	535	28.1	6.37e-3	300	6.4e-6	1000
ethylbenzene	0.87	(31.48-0.11*T)	152	7.00	6.43e-3	1100	7.8e-6	700
xylenes †	0.87	(31.48-0.11*T)	168	8.9	7.6e-3	894	7.9e-6	10,000

*from Simms et al. (1991) and Mercer and Cohen (1990), except ** (USEPA from Fetter (1993) and DeZuane (1990)).*

† *average of m-xylene, o-xylene, p-xylene values*

Table 3.10: Contaminant properties for Bay Area site

Contaminants	S_G	σ_{IFT}	S_o	vp	H	(K_{oc})	ν	MCL **
		(dynes /cm)	(mg/l)	(mm Hg)	(atm m^3/mol)	(ml/g)	(m^2/s)	(ug/l)
TCE	1.47	29.5	1100	57.9	9.10e-3	126	3.9e-6	5
xylenes †	0.87	(31.48-0.11*T)	168	8.9	7.6e-3	894	7.9e-6	10,000

*from Simms et al. (1991) and Mercer and Cohen (1990), except ** (USEPA from Fetter (1993) and DeZuane (1990)).*

† *average of m-xylene, o-xylene, p-xylene values*

Table 3.11: Contaminant properties for Verona Well Field site

	S_G	σ_{IFT}	S_o	vp	H	(K_{oc})	ν	MCL **
Contaminants		(dynes /cm)	(mg/l)	(mm Hg)	(atm m^3/mol)	(ml/g)	(m^2/s)	(ug/l)
1.1.1-TCA	1.35	(28.28-0.12*T)	1500	123	1.44e-2	152	6.7e-6	200
TCE	1.47	29.5	1100	57.9	9.10e-3	126	3.9e-6	5
PCE	1.63	32.86	150	17.8	2.59e-2	364	1.2e-5	5
1.2-DCA	1.26	(35.43-0.14*T)	8520	64.0	9.78e-4	14.0	7.0e-6	5

*from Simms et al. (1991) and Mercer and Cohen (1990), except ** (USEPA from Fetter (1993) and DeZuane (1990)).*

Table 3.12: Contaminant properties for Atlantic Canada Airport site

	S_G	σ_{IFT}	S_o	vp †	H	(K_{oc})	ν †	MCL **
Contaminants		(dynes /cm)	(mg/l)	(mm Hg)	(atm m^3/mol)	(ml/g)	(m^2/s)	(ug/l)
JP4 (jet fuel)	0.81 to 0.82	25	60	60	-	-	1.3e-6	-

*from Simms et al. (1991), Mercer and Cohen (1990) and Kirk-Othmer (1978), except ** (USEPA from Fetter (1993) and DeZuane (1990)).*

† *Assumes properties similar to ASTM Jet Fuel B.*

Table 3.13: Summary of Case Studies

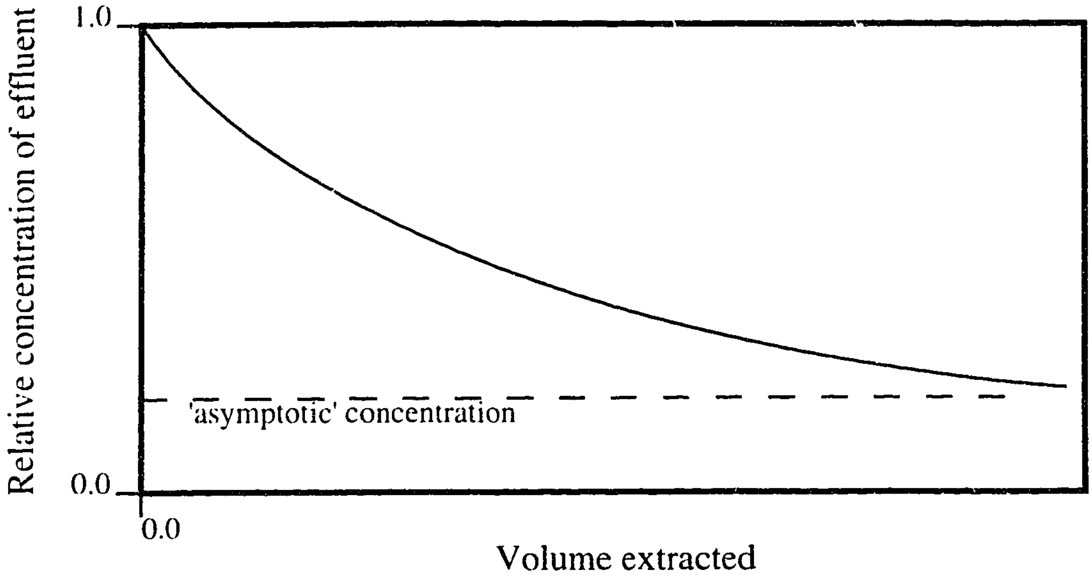
Case	Contaminants of Concern	Aquifer Geology	Plume Area	Plume Pore Vol. (est.)	Pore Vol.'s Extracted	Area Reduction
			(m^2)	(m^3)		
South Brunswick Township	1,1,1-TCA; PCE; TCE; Toluene; Benzene	fluvial sand and gravel	640,000	2,600,000	0.1	70 %
Central New Jersey	TCE; 1,1-DCE; Cis-1,2-DCE; Trans-1,2-DCE; Vinyl Chloride	marine sand and silt	9,500	8,700	2.7	- 260 % (area increase)
Firestone	1,1-DCE; 1,2-DCA; 1,1-DCA; TCE; PCE; 1,1,1-TCA; Toluene; Xylenes; Ethylbenzene; Benzene	alluvial sand and gravel channel deposits	300,000 (shallow), 630,000 (intermediate), 945,000 (deep)	1,530,000; 2,270,000; 7,100,000 (respectively) 10,900,000 (total)	0.5	100 %
Woodland Park, Michigan	1,1-DCE; phenol	Sand and silty sand, glacial outwash	58,000	230,000	15	98 %
Strasbourg-Entzheim Airport †	kerosene (aviation turbo fuel TR4)	Rhine alluvial deposits	400,000	600,000 (3000-6000 free product)	not enough data	2-4 %
Southern Ontario A	gasoline (BTEX)	fractured sandy, silt till	4,000	15,000 soil (15.1 free product)	0.20	150 %
Southern Ontario B	gasoline (BTEX)	coarse granular channel in glacial outwash	not given, up to 350m dissolved plume length in channel	not enough data	18,000 m^3 ground water extracted	not enough data
Bay Area	TCE; xylenes	inter-bedded sands, silts gravels and clays	3,000	1350 (2800 with aquitard)	2.1 (1.0 with aquitard)	85 % avg. conc. reduction

Table 3.13: Summary of Case Studies (continued)

Case	Contaminants of Concern	Aquifer Geology	Plume Area	Plume Pore Vol. (est.)	Pore Vol.'s Extracted	Area Reduction
			(m^2)	(m^3)		
Verona Well Field	1,1,1-TCA; TCE; PCE; 1,2-DCA	fractured sandstone	not given	not enough data	not enough data	containment objective achieved
New England Service Station	automobile gasoline (BTEX)	high yield glacial aquifer	1400	6,400	1.5 to 3.0	50 % (90 % dissolved conc. reduction)
Coastal airport †	jet fuel (JP4)	high yield sand and gravel (man-made)	4,700 (free product)	450 (free product spill volume)	not given	(16 % free product volume extracted)

† *free product extraction only, ground water pumping for containment.*

Figure 3.1: Typical Asymptotic Response to Pump-and-Treat Remediation



CHAPTER 4: CHARACTERIZATION AND DESIGN USING SCIENTIFIC AND EMPIRICAL DATA

4.1 INTRODUCTION

A beneficial perspective can be gained from analyzing the scientific aspects of the contaminant transport regime, with respect to pump-and-treat remediation, and examples of field experiences. This could be accomplished through linking the flowchart associated with the contaminant hydrogeology regime (Figure 2.1) and a flowchart illustrating the process of remedial system evaluation, design and operation (Figure 4.1). Figure 4.1 should be considered a summary of the remediation methodology applied in varying degrees to the case studies of Chapter 3.

The discussion of various methods for property and process definition in Chapter 2 used terms such as direct characterization, inference, and prediction. Direct characterization referred to various analysis techniques that have been used to determine properties and processes in a relatively direct manner. It does not, therefore, differ qualitatively from the much more complex procedures of inference and prediction. However, the distinction is important in indicating the level of analysis required for evaluation of the parameter.

The term 'prediction' should be used with caution due to potentially misleading interpretations. It should not be taken to refer to estimates of actual future performance. Rather, it is better applied as a tool to optimize design configurations. The accuracy of parameter definition, process definition and calibration actions (through direct characterization and/or inference) therefore dictate the transferability of the optimized design configuration to the field; the closer the model input parameters are to reality, the closer the observed performance of the design comes to the predicted optimal. Performance 'predictions' through modelling should therefore be applied with caution in pump-and-treat operations.

Inference should be used as an analysis tool to evaluate various parameters in a complex regime. Refinement of parameter values and conceptualized boundary conditions (in a numerical or analytical model) could then be performed to provide a more accurate input for optimization, again using prediction methods.

This chapter will provide an analysis of the relationships between the two perspectives of the contaminant transport regime stated previously. It should not be forgotten that the central focus of this discussion is to relate these aspects of contaminant hydrogeology and remedial system design to pump-and-treat remediation. A discussion of numerical modelling application to inference and prediction will then be undertaken. This will be followed by an example of modelling using one of the case studies introduced in Chapter 3.

4.1.1 Remediation System Evaluation, Design, and Operation

The flowchart presented in Figure 4.1 illustrates a generalized methodology for remediation of contaminated sites. This methodology follows from identification of the problem to operation closure. A brief description of the components of this flowchart and their relationship with other components follows.

Upon identification of a contamination problem, three applicable areas need to be defined. First, the site must be physically and chemically characterized to the most detailed level practical. Second, the contaminant(s) themselves must be identified and characterized. Third, the governing regulatory environment must be defined, including all levels of government with jurisdiction over the site, contaminant and operation. The Firestone Site in Salinas, California provided an example of extensive investigation and characterization of both site and contaminant prior to remediation (Smedes *et al.*, 1993). As a Superfund site, the regulatory environment was clearly defined at the outset.

The initial stages of the characterization are often used to determine the need for, and subsequently to design and implement, interim actions. These actions are critical in minimizing the future extent of contamination to be remediated by the full scale operation. There is also a feedback loop between site characterization and interim action which includes the monitoring system. Monitored responses to interim action often provide data that supplements initial (and future) site characterization efforts. Monitoring and sampling are also closely linked to the identification and characterization of the various contaminants in the subsurface. The reduced pumping of public supply well SB11 at the South Brunswick Township site was a good example of quick interim action designed to protect potential receptors until detailed investigation and remedial action could be undertaken (Roux and Althoff, 1980).

The risk analysis component is singled out on the flowchart as an optional step between the characterization stage and the design process. In some regulatory environments, the required actions or acceptable levels of contamination are fully defined by regulatory agencies. However, other jurisdictions may require risk analyses to be performed for each specific case, if conservative screening criteria are not met. The results of these analyses, or the aforementioned dictates of the governing regulatory agency, lead to the identification of goals for the remediation project. The process of setting these goals requires input from the risk analysis and perhaps from the various characterizations of the site. Clearly, these goals should be set at attainable levels that match the realities of the project.

It is an important point that many jurisdictions use general regulatory standards to select goals with limited input from site specific characterization. The process of a site specific risk analysis would take into account toxicity, mobility and persistence of the contaminant with respect to the site's physical and hydrogeologic layout and the proximity of receptors. Use of general standards might require that the worst case configuration be used when setting acceptable levels. This could lead to goals that would be excessive and possibly unattainable, given existing site conditions. The Firestone site showed an example where stringent clean-up standards were imposed by the governing regulatory agency but subsequent risk analysis indicated that almost no significant risk to receptors was possible (Smedes *et al.*, 1993).

Evaluation of the various remediation systems needs input from the site, regulatory environment, public perception, contaminant and project goals. This eventually leads to selection of the system to be implemented and its design. Donovan and Murray (1985) described an evaluation process that encompassed considerations of hydrogeologic properties, hydrostratigraphy, contaminant properties and regulatory requirements. The design of the chosen system requires iterative input from the site, the contaminant characterizations and the goals. It should be noted that, as the design process progresses from goal identification to system selection and design, the level of detail required from the characterization stage becomes greater. Following design and construction (or installation), the system is implemented and operated for its design life. Monitoring again becomes an important input component at this stage. As the operation continues, monitoring results may lead to further design modifications and eventually to closure, depending on the nature of the monitored responses. The iterative relationship between system design and operation illustrated in the flowchart suggests this continuing process. Again, the Firestone site case study provides an excellent example of this iterative approach (Smedes *et al.*, 1993). Final closure usually requires a period of extended monitoring beyond the shutdown of active remediation, as described by Westerby (1992) for Southern Ontario site B.

Though it is not indicated directly by the flowchart in Figure 4.1 permitting is usually required before any remediation action may be implemented. This permitting process through the regulatory agency is needed to install and operate the remediation system or to handle the by-products of the operation. It must therefore be considered during the evaluation and design phases.

4.1.2 Interaction of Scientific and Empirical Perspectives

Scientific and empirical knowledge interact and shape the pump-and-treat remediation system design and operation. A comparison can best be achieved by relating the components of the two flowcharts (Figures 2.1 and 4.1) and their various interactions. The relationship between these two perspectives could be used to expand understanding of each for a given site.

The characterization process indicated in Figure 4.1 encompasses many of the components of Figure 2.1. However these elements are investigated to varying levels, with some having immediate delineation and others requiring input from other elements of the remediation methodology. Site characterization usually encompasses elements of the solid matrix and fluid phase categories. Commonly geology, topography, geotechnical and hydrogeologic properties, and climate are initially investigated. This subsequently leads to definition of the hydrogeology, local recharge/discharge rates, and the regional hydrogeology. Contaminant identification deals with those elements falling into the contaminant category of Figure 2.1. This largely refers to sampled investigation of the site chemistry and geochemistry and site records of chemical handling and inventory.

Further investigation, with possible input from interim actions, often leads to a greater resolution of the various components of Figure 2.1. This is usually accomplished with the inference and prediction analysis techniques mentioned in Chapter 2 and described in more detail later in this chapter. The definition of solute and NAPL transport components is often added at this juncture. Also, various attenuating processes in both the physical and chemical regime should be defined here.

The legend in Figure 2.1 indicates that direct investigation can be used (with varying degrees of applicability) to define each of the elements presented. It also shows the areas where inference and prediction could be applied between components. This might be accomplished with analytical or numerical techniques as described in Chapter 2. It is important to note that these analysis techniques often use responses to interim or site characterization actions (such as pumping tests) as input. The responses are measured through monitoring and sampling actions and processed for input to the analysis.

Monitoring and sampling are used to define the hydrogeology, chemistry, and occasionally geochemistry. As the local recharge/discharge regime changes, in response to added stress from the remediation action or pumping tests, the monitored hydrogeology also changes.

This response can subsequently be used to infer additional information about the hydrogeologic properties and regional hydrogeology. The response of solute and NAPL transport to the changed ground water regime could also be used to infer transport parameters that would otherwise be difficult or impossible to determine. This would be analogous to tracer tests used in direct characterization, although these tests would use full scale field responses as input. The example presented later in this chapter provides a good illustration of these procedures.

Following characterization of the site, contaminant, and regulatory environment, there is a need to define the goals of the remediation project. As mentioned previously, risk analysis is sometimes required on a site specific basis, depending on the regulatory environment. When this is the case, the proximity of receptors and their likely exposure levels over time is a primary issue. The solute and NAPL transport properties of the contaminant play a large role in defining this exposure risk. Attenuating actions usually have beneficial influence on this risk but may be neglected in order to produce a conservative prediction. When contaminant levels are specified by the governing regulatory agencies, the selection of goals can be straightforward and little input from the scientific regime is necessary.

With goals for the remediation project defined, the process of system design and operation begins. Evaluation of the various available remediation systems requires input from the three areas of characterization. Different remediation methods often have differing critical components in the contaminant hydrogeology regime of Figure 2.1. Pump-and-treat techniques, for example, rely primarily on control of the ground water flow regime and thus relate largely with the hydrogeologic properties and existing hydrostratigraphy. Other methods, such as vapour extraction and air sparging, rely on the chemical properties of the contaminant more than other components. The final selection will be influenced by various considerations in the physical and chemical world, as well as the political considerations of the regulatory world.

Monitoring and sampling actions usually drive the iterative design and operation process suggested in Figure 4.1. These actions revolve around the hydrogeology, chemistry, solute transport and NAPL transport components of contaminant hydrogeology.

Closure is heavily dependent upon monitoring results and often extends beyond the active period of remediation. At the point of closure the subsurface regime is as well defined as it is likely to become. Therefore, close observation of the chemistry is the dominant factor in closure decisions. The decision for closure is usually made by the regulatory agency at the request of the remediation project operators or owners.

4.2 INFERENCE AND PREDICTION METHODS

The number of interactions and interdependencies in the contaminant hydrogeology regime, illustrated in Figure 2.1, can result in an intractable problem when attempting to model certain processes. The simpler components can be modelled with straightforward analytical methods but more complex processes require numerical methods. Inference and prediction are terms referring to the procedure of modelling these actions and interactions in some form or another. A discussion of the various methods of using inference and prediction to characterize the contaminant transport regime will now be undertaken. A comparison of inference and prediction to direct determination methods will be presented, followed by a listing of available model types for use in various areas of the regime. Some innovative models that incorporate stochastic theory, coupled analysis, and dual porosity will also be reviewed.

4.2.1 Inference and Prediction versus Direct Determination

The flowchart presented in Figure 2.1 indicated that each element might be characterized through direct determination, while some could also be defined through inference or prediction. Methods of direct determination vary depending on the nature of the property or process to be defined. For example, grain size distribution (a geotechnical property) could be determined through a laboratory sieve analysis of borehole samples while pore water chemistry might be determined in-situ through ion specific conductivity or ex-situ through laboratory analysis of samples. Some components of the regime are almost impossible to determine directly. Geomorphology, for example, would more likely be inferred through interpretation (often subjective) of numerous other components.

Direct determination might refer to methods of inference that are relatively simple and well understood. An example of this would be in-situ hydraulic conductivity tests using pore pressure dissipation behaviour. The observed pore pressure curves are used to infer local hydraulic conductivity through some analytical solution of the assumed flow field. The absolute definition of direct determination becomes somewhat vague when this is considered. For the sake of simplicity, it will be considered to be analysis which requires only partial input from another component of the regime, rather than fully detailed data. In the example of in-situ hydraulic conductivity testing, pore pressure distribution around the sensing device could be defined as hydrogeology, but not of sufficient scope to influence the hydrogeology of the entire site significantly.

Given the above definition of direct determination, inference and prediction must be taken as the case where broad data input from other components is required to define the process or property in question. This is most often applied to fields where interdependency of various elements leads to non-linear relationships. The nature of these relationships dictate whether analytical or numerical solutions are necessary to find the solution.

4.2.2 Various Inference/Prediction Models

The term inference has been used quite often in this chapter to refer to the determination of a property based on the behaviour of processes dependent on the values and distributions of that property. In section 2.6.4 that process was also described as one method of inverse modelling. In this type of inverse modelling, solutions to the governing equations for a process would be generated for assumed values and distributions of various properties. These solutions would be compared to observed values for dependent variables and a residual subsequently calculated. Modification of the input to the model could be then performed in some systematic manner until the residual was minimized and of an acceptable value. This residual might be considered globally based on summed calculations, or locally within the model area based on qualitative interpretation of the solution plot.

One common application of inverse modelling methodology is in ground water flow field simulation. The dependent variables (hydraulic head) at observation wells would be compared to calculated values to obtain a residual field. Through modification of the material properties, boundary conditions, and possibly the discretization, a close match may be obtained. Another area of inverse modelling is with regard to contaminant transport, where solute migration through the subsurface is often modelled as an extension to the ground water flow field definition. In this case, the dependent variable (solute concentration) is matched to observed values while material properties such as dispersivity, retardation coefficients, and decay processes are adjusted.

Errors in the flow field definition would subsequently be transferred to the contaminant transport model through coupling, usually via an inaccurate pore-water velocity field. The flow model may seem to be initially calibrated under natural stresses. However, changes in the stress field applied to the site might bring out inaccuracies in the model through erroneous responses to the new stresses. This inaccuracy would then lead to errors in the advection driven contaminant transport model response.

It is in this area of contaminant transport modelling where many of the difficulties arise with respect to characterization. The complexity of Figure 2.1 illustrated many interactions that occur between components of the regime. If each individual component is also spatially (and even sometimes temporally) variable then the problem can become intractable. A case in point is the issue of solute transport in a geochemically active media. Some processes to be considered in this scenario include: partitioning of mass between the aqueous and adsorbed solid phases; chemical speciation of the aqueous phase among various compounds and complexes; distribution of the solute between NAPL and water, etc. Heterogeneity of these characteristics within a site multiplies the complexity. Rate effects (kinetics) further excaberate the problem. Finally, all this must be added to the complexity of the hydrostratigraphy and the accuracy of its modelled solution.

The above discussion illustrates the main limitations of inverse modelling of solute transport. The independent variable, or material properties (and rate effects), are so numerous that the solution set which minimizes the residual is often not unique (Peck *et al.*, 1988). Also, the system of modifying these variables to obtain a solution cannot easily be predefined. To simplify the problem, many of the interacting processes are often assumed inactive, negligible or constant (spatially and temporally) during modelling while solving for others. Kinetics are almost always neglected, geochemical properties are assumed to be homogeneously distributed or at least zoned and constant with time, chemical reactivity effects are usually neglected in favour of single species analysis. The above simplifications are not always ignored. They may be investigated individually while excluding other influences in order to confirm the acceptability of various assumptions. However, using this simplification process as a solution to the problem may be scientifically unacceptable.

Numerous individual commercial models are available for various areas of the contaminant hydrogeology regime. The extensive list and summary presented in Table 4.1 gives a partial picture of the variety of models in use, particularly in ground water flow and contaminant transport. Many of these models have direct application to pump-and-treat actions (eg. ground water flow, contaminant transport, parameter estimation); others might be useful in more completely characterizing the processes acting on a site, or in performing sensitivity analyses on the various processes neglected in the flow and transport models. This list is a summary of information taken largely from Internet resources and is supplemented with some individual product material. The Internet source was a series of on-line software catalogs from the International Ground Water Modelling Center (IGWMC). The extent of this table gives some indication of the variety of models available specifically for ground water related issues.

The important areas addressed by ground water flow and solute transport packages in Table 4.1 are steady-state or transient ground water flow; transport of dissolved, vapour phase and/or immiscible phase chemicals; saturated, unsaturated, saturated-unsaturated, water-oil, and three phase (air-water-oil) media. These programs include capability in one, two (area and/or section), and three dimensions as well as in axisymmetric regimes. The media might be homogeneous or heterogeneous, isotropic or anisotropic, and porous or fractured. The transport models are commonly capable of handling advection, dispersion/diffusion, linear or nonlinear adsorption, ion exchange, and first-order decay processes. Some of the more specialized features available through these flow and transport models are: stochastic head distributions (WHPA), stochastic concentration distributions (USGS-SOL), parameter estimation for hydraulic conductivity and recharge (FTWORK), variable density flow (SWICHA or MOCDENSE), dual porosity transport (TRAFRAP-WT or BAREN), coupled hydrocarbon and oxygen transport (BIOPLUME II), viral particle transport (CANVAS or VIRALT), multiple species transport (MOCDENSE), decay product transport (PRZM), immiscible fluid infiltration (SUMATRA-1), volatilization and gas diffusion (VLEACH).

There are also several specialized packages for estimation of parameters for ground water flow, solute transport, and unsaturated flow. Some of these models are designed specifically to analyze test data for 'direct characterization' methods such as pump tests or tracer tests. Others might be used to calibrate full scale models through modification of parameters with respect to residual values.

Several geostatistical packages are also presented. These programs could be used to calculate statistical parameters from spatial data. Alternatively, they might be used to generate randomly distributed property fields for use in Monte Carlo analyses. A few models also determine optimal locations and design values for pumping/injection or monitoring wells (MAP/MEMO or MODMAN). MODMAN is a post-MODFLOW package for optimization of design. Other post-MODFLOW packages are commonly available for capture zone analysis or water budget determination.

Hydro(geo)chemistry models are also presented. These include models for mass-balance reactions, ch[illegible]librium speciation, geochemical equilibrium and non-equilibrium speciation.

4.2.3 Coupling of Models

The preceding section and its extensive software listing (Table 4.1) described various models that involved solutions to single processes such as ground water flow. Some of these programs were capable of coupling with completely separate model types through external data files. An example of this is saturated ground water flow and solute transport (for example MODFLOW and MT3D). Coupling of these models is usually accomplished through a data file of ground water velocity fields. Other models couple two or more separate internal solvers. Examples of these are coupled oil-water flow (ARMOS and MOTRANS), coupled air-oil-water flow (MOTRANS), coupled multiphase flow and solute transport (SPILLTRANS), and coupled multi-species transport (BIOPLUME II, SPILLTRANS, and MOCDENSE). Dual porosity models for fracture and matrix solute transport (TRAFRAP-WT and BAREN) are usually coupled through the boundary concentrations between the fracture fluid and the matrix pore fluid.

One area where coupling capability has historically been lacking is between solute (and/or NAPL) transport and hydro(geo)chemistry. The common simplifying assumptions of single species transport, instantaneous-reversible linear adsorption, and first-order decay have been discussed previously in this study. Multiple species transport models seldom include chemical speciation between various compounds, complexes or ion pairings. Also, the concentrations used for partitioning are assumed to be total concentrations with activity coefficients less than unity neglected. Non-equilibrium effects have also been commonly neglected in transport packages, though one vapour transport program does accommodate it through an efficiency factor for adsorption (VENTING). Brusseau (1990) provided an extensive examination of the limiting effects in operation during dissolved and immiscible phase contaminant transport.

The benefits of coupling all significant processes into one integrated model must be considered critically. It is obvious that a set of existing programs could be integrated today to create one all encompassing model. However, full understanding of the driving forces behind the numerous geochemical processes is still lacking (Brusseau, 1990). The increase in model realism would be offset by dramatically increased data requirements and greatly decreased computational efficiency. Therefore, a fully coupled model of the contaminant transport regime may be attainable but its value would be questionable at present. However, with further improvements in computer power, programming efficiency and versatility this constraint may be overcome.

4.3 MODELLING APPLICATION TO A CASE STUDY

The following section will apply one of the more basic flow and solute transport model couplings to a published pump-and-treat case history. MODFLOW and MT3D are three-dimensional finite difference models for ground water flow and solute transport, respectively. As outlined in Table 4.1, the MODFLOW package is capable of transient flow, heterogeneity, anisotropy, irregular geometry, and layered semi-confined or unconfined aquifers. MT3D can model the advection/dispersion equation with sink/source mixing, linear or nonlinear adsorption, first-order decay, and confined, variably confined or unconfined aquifers. These two models are quasi-coupled in a stepwise fashion through an unformatted head and flow file, output by a modified version of the MODFLOW program specifically for input to MT3D. That is, for simultaneous transient analysis of flow and transport the transient flow field is saved as a sequence of steady-state flow fields for use in the transient transport model.

This modelling effort was undertaken to examine the amount of detail in process definition that might be derived from limited initial information, using inference through numerical modelling. Data sources supplemental to the case study papers were used as outlined in section 3.2. Several necessary assumptions, and those inherent in the applied models, were applied as required. The degree of data available for this model might be postulated to be representative of an initial characterization attempt in the early stages of investigation. Data uncertainties and inadequacies will be discussed as they are encountered.

One critical limitation of the type of solution forthcoming from finite difference models is artificially induced or enhanced transport along grid and column lines. This characteristic of finite difference models is largely due to the lack of cross-terms in their solution equations. Warwick and Stoffregen (1991) investigated the operation of this effect with respect to the U.S.G.S. MOC model and concluded that significant numerical dispersion could occur if the grid orientation was not orthogonal to the modelled flow field. Due to this limitation, discretization spacing and orientation is critically important in constructing a useable model for finite difference programs. In the following case care was taken to align the grid as closely as was practical to the dominant transport direction. Some difficulty was experienced in this respect due to the curvilinear flow field.

4.3.1 Model Details

The study modelled in this section was the South Brunswick Township case described in section 3.2, Chapter 2. This case was chosen for its relatively extensive supply of information regarding site stratigraphy, hydrogeologic properties, hydrostratigraphy, and plume delineation both before and after pump-and-treat action. The data used in this model was largely limited to that supplied by two papers; Roux and Althoff (1980) and Althoff *et al.* (1981). Critical gaps in the supplied knowledge were filled through supplemental sources and assumptions as described here.

The site was modelled in two dimensions (in plan) on a rectangular 39 by 46 cell (4600 by 5600 feet) grid encompassing the entire initial estimated plume area. Imperial units were used in the modelling effort to aid in comparing values and results to the original case study. One-hundred and sixty-three cells were deactivated in the upper and lower right corners of the grid in order to follow the irregular boundaries of the modelled area. This left a total of 1631 active cells in a single layer, varying in either horizontal dimension from 50 to 200 feet. The finer grid spacing of 50 by 50 feet was centered over the postulated contamination source in order to accommodate the high concentration gradients and pumped hydraulic gradients to be modelled there, as well as to minimize the finite difference dispersion error discussed previously. Though both MODFLOW and MT3D were capable of modelling multiple layer systems, the data supplied in the above papers was not sufficient to define the lower aquifer or to infer it through responses to hydraulic stresses.

Figures 4.2, 4.5 and 4.8 give schematics of the block centered finite difference grid layout of the site used for all flow and transport modelling. Buildings and various monitoring wells were shown in these figures with respect to position, orientation and scale. The left side of the grid was oriented approximately 37 west of north as indicated by the arrow on the figures. The relative position and orientation of the various emplacements was derived from digital scans of the drawings supplied in the two previously referenced papers. These scans were imported directly into the drafting program used to produce the figures. Consequently, the accuracy of the positioning on these emplacements was limited by the accuracy and resolution of the original figures. It was noted during this transfer procedure that variation in position of some emplacements was noticeable between supplied figures. Given the resolution and consistency of these figures, the size of the monitoring well symbols is somewhat indicative of the confidence given to their positioning within the grid. The observed water-table and concentration contours were overlaid on these figures for comparison to model results.

The MODFLOW and MT3D programs used were both capable of modelling variable thickness in aquifer layers. To take advantage of this feature an attempt was made to contour the bottom of the model layer (or the top of the Woodbridge clay aquitard) to roughly reflect the water-table topography, as described in Roux and Althoff (1980). However a drastic reduction in computational efficiency resulted from this attempt, due to the increased number of calculations necessary to solve the matrix equation. This increase hampered the application of iterative steps necessary for an inverse modelling procedure. A single model run that reached a solution in under one hour (on an IBM RISC 6000 computer) under constant aquifer thickness took roughly 24 to 48 hours with the variable thickness configuration. Numerical dispersion and solution stability problems were also experienced in this configuration. Run solutions contained many cells with unrealistically large concentration values (greater than 10^{49}) around the boundary of the grid. Since the origin of this aquifer geometry was largely conjecture, it was decided that little could be gained by pursuing this problematic path. A constant thickness aquifer was therefore used in this modelling effort.

It is important to note that the water-table configuration indicated on Figure 4.2 was not the same as that under which the majority of contaminant transport took place. Roux and Althoff (1980) indicated that the discharge from public supply well SB11 was reduced to 0.4 million gallons per day (gpd) in order to prevent further migration toward the nearby aquitard 'window' (no previous discharge rate was given). When comparing the given water-table (Figure 4.2) with the configuration of the original contaminant plume (Figure 4.3) it became evident that an incompatibility existed between the two. Due to a lack of information on the historic hydrogeology, several important assumptions were made in order to create a ground water regime that would duplicate the original migration pattern indicated in the study.

Since the water-table configuration was not known for the original transport phase, inverse modelling of the upper aquifer for calibration of the hydrogeology was undertaken for the supplied water-table configuration. The modelling was therefore done for three stages, numbered chronologically: rough ground water modelling of the historical transport phase hydrogeology and subsequent plume development to match given data (stage 1), calibration modelling of the hydrostratigraphy based on supplied hydrogeology during the reduced pumping stage (stage 2), and finally modelling of flow and transport for the initial stages of the pump-and-treat system operation (stage 3).

4.3.2 Modelling Prior to Pump-and-Treat Operation

There were two stages that occurred prior to pump-and-treat remediation in this case. Stage one might be considered the natural condition of the site over the past few decades, while stage two would represent the site's response to the interim action of reduced (SB11) pumping. The initial modelling of the contaminant plume development required a very rough water-table estimation. Consequently, this model was used only to develop the required initial plume with very limited calibration performed on the flow or transport parameters. Second stage modelling involved calibration of the site's hydrogeology based on matching the ground water model to the given water-table configuration. The model was calibrated to this second stage first in order to increase confidence in the hydrogeology for input to the first stage.

4.3.2.1 Hydrogeology Calibration in Stage 2

Calibration of the hydrogeologic parameter distribution for the site was performed based on the supplied water-table configuration. As mentioned above, this regime was influenced by a reduction in SB11 pumping rate prior to the given monitoring well readings. An assumed steady-state analysis was performed to obtain the flow field for this stage. The data input files for MODFLOW are listed in Table B.1 (Appendix B). Observed values for the water-table elevations at monitoring wells were approximated in the isopotential contours shown in Figure 4.2, and are listed in detail in Table 4.2.

The MODFLOW input data listed in Table B.1 described a steady-state ground water flow model using the basic data input file (BAS) and the following modules: block centered flow (BCF), well operation (WEL), recharge (RCH), strongly implicit procedure (SIP), and output control (OPC). The model was run with starting head values approximated from the water-table contours shown in the published studies. Table B.2 gives a complete listing of these starting heads for the model grid with the constant head cells highlighted in italics.

A zero flow (Neumann) boundary condition followed the left (northwest) side of the grid with specified head (Dirichlet) boundaries along the remaining sides. The aquitard 'windows' were modelled as groups of specified head cells as well. It would have been desirable to extend the ground water flow model farther from the site and assign more realistic boundary conditions. However, limitations in the available hydrostratigraphy and hydrogeology data dictated a more forced approach for this study.

The hydraulic conductivities of the cells in the single layer were assigned in zones. These values were varied within the operable range suggested in Roux and Althoff (1980). The hydraulic conductivity varied in three stages from 50 feet per day (fpd) at the top portion of the grid to 350 fpd for the bottom half as indicated in the SBT2BCF.DAT input file of Table B.1, Appendix B. Recharge at the site was set at 0.0005 fpd through the iterative calibration procedure.

The water-table configuration that was produced at the end of stage two model calibration is presented in Figure 4.3 for qualitative comparison to the given configuration of Figure 4.2. Table 4.2 shows the residual calculations using the model output values.

Determination of the residual for this stage consisted of calculating the absolute value of the difference between supplied monitoring well values and values calculated from the model output. These model output values were calculated for each of the 36 wells indicated on Figure 4.2, with the head value taken as the single cell value, two cell average, or four cell average depending on the position of the well with respect to the model grid. The well locations indicated on the table were estimated to the nearest half-cell dimension, based on figures in Roux and Althoff (1980) that were scanned into the drawing of the model grid. A spreadsheet program performed these calculations using imported model output data. Table 4.2 shows a sample of this spreadsheet with the last residual calculation results for stage 2, hydrogeology calibration.

The manner of cell value averaging for residual calculation described above does have some shortcomings in accuracy, especially in areas with high head or concentration gradients. However, this was a general example of a modelling investigation and the number in question was simply used as a relative value, taken to minimize the average residual of the grid. The inaccuracies of the residual calculation could be said to balance themselves to a sufficient degree ıor the application to which these numbers are put.

A short sensitivity analysis was performed on the model for the aquifer's hydraulic conductivity zones. An average conductivity value of 250 fpd was distributed across the entire grid. The resulting hydrogeology was presented in Figure 4.4 for comparison and discussion.

4.3.2.2 Plume Development Modelling in Stage 1

Though the time period of original contaminant transport (stage 1) largely preceded the reduced pumping stage (stage 2), calibration of the hydrogeology was performed for stage

2 prior to plume development modelling as described above. The modelling procedure for plume development involved transient contaminant transport analysis under a steady-state flow field. MT3D input data files for the contaminant transport modelling of stage 1 are listed in Table C.1, Appendix C. MODFLOW input data files for this stage were identical to those calibrated files of Table B.1 with the few changes listed in the following paragraph. Figure 4.5 illustrates the observed stage 1 contaminant plume, adapted from Roux and Althoff (1980), positioned with respect to the site and monitoring well locations as well as the grid layout and orientation. The numbering system of the monitoring wells for solute concentrations is the same as that used in the previous ground water modelling stage. However, some new concentration monitoring wells were placed on this figure, and some previously used water-table monitoring wells were omitted. Table 4.3 shows the monitored concentration values for each well shown in Figure 4.5.

The initial contaminant plume was modelled using a ground water regime created by one major change from stage 2 conditions. This was the assumption that the pumping of SB11 affected the upper aquifer only through the aquitard 'windows'. Therefore, the higher pumping rate alluded to in the published studies was approximated simply by reducing the constant head values of the cells over this window. The western opening (nearer to SB11) was reduced from approximately 80 to 70 feet (see italicized values in Table B.2). This produced a ground water regime that would transport solute along the direction indicated in Roux and Althoff (1980) and Althoff *et al.* (1981); that is, along the grid column orientation in Figure 4.5. Some limited calibration of dispersion parameters was done in an attempt to match the spacing of isoconcentration lines and minimize the plume residuals in the results. However, given the amount of conjecture involved in developing this flow regime, calibration efforts were constrained to a few probable values. Since the length of time over which contamination occurred was not given, other parameter calibrations were not possible (i.e. retardation factor).

The grid configuration of the stage 1 model for the MT3D program was identical to that used in stage two flow model calibration. The elevation of the top of the layer was input as 95 feet and the cell thickness was therefore 40 feet (the difference between this value and the bottom elevation input for MODFLOW, as described previously). Sensitivity of the transport phase to the averaged hydraulic conductivity model solution was investigated and will be discussed later. A porosity of 40% was assumed for the entire grid.

Boundary conditions used for the contaminant transport model were specified advective-dispersive flux (Cauchy boundary condition) at all grid boundaries, and a specified concentration (Dirichlet boundary) at the contamination source. The imposed conditions from these boundaries was not expected to have a significant influence on the solute plume, provided that a sufficient distance between the two was maintained for the entire modelled time period.

The length of time for contaminant transport to create a plume that closely matched that observed in the monitoring wells was found to be approximately 10 years. The modified method of characteristics (MMOC) was used in modelling transport for this study. Upstream finite difference modelling was used for some of the initial model runs in order to roughly match the plume configuration before applying the more rigorous, but less computationally efficient, MMOC solver. A Courant number of 0.75 was set and 16 particles were specified for sink cells in the MMOC solver.

The longitudinal dispersivity was set at three feet with horizontal and vertical transverse dispersivity set at 10% and 1% of this value, respectively. These numbers were taken from Tables A.2 and A.3 (Appendix A) as typical values for the geology of the area. Molecular diffusion was neglected in this model due to the high ground water flow rates involved. Recharge was set at zero solute concentration across the site. One point source of 4,000,000 ppb solute concentration was specified at the approximate center of the plant site, roughly the aqueous solubility of 1,1,1-TCA Chemical reactions were assumed to follow a linear adsorption isotherm with no decay. The entire grid was input as geochemically homogeneous with 103 lb/ft^3 bulk density, and K_d equal to 0.003 ft^3/lb. The basis of these chemical and geochemical parameter estimates was Table 3.2 and its sources.

Figure 4.6 illustrates the modelled plume that resulted from the above configuration. The residuals for the concentration plume were calculated in a similar manner to that used in the previous section for the hydraulic head distribution. Table 4.3 gives a sample of the spreadsheet used for calculation, with the final residual calculation shown. The modelled isoconcentration distribution (Figure 4.6) was also checked visually for qualitative similarity to the given solute plume of stage 1 (Figure 4.5). Figure 4.7 illustrates model results for stage 1 plume development given the simplified hydrogeology of the stage 2 sensitivity analysis (homogeneous hydraulic conductivity).

4.3.3 Pump-and-Treat System Operation in Stage Three

The data available for the changed flow regime of stage three was extremely limited. The location and discharge rate of the seven pumping wells was specified satisfactorily.

However, the hydraulic response of the aquifer was not described. Consequently, the previous calibration actions were relied upon to have created a flow regime that would respond realistically to pumping for the withdrawal transport stage. This stage was modelled using transient analysis for both flow and transport. As described earlier, MT3D uses steady-state flow regimes in a stepwise fashion to couple simultaneous transient flow and transport analyses. Input files differing from those used in stage 2 for MODFLOW and stage 1 for MT3D are listed in Table D.1, Appendix D.

The initial and final plume layouts, after ten months of pump-and-treat system operation, were supplied by Althoff *et al.* (1981). These have been reproduced in Figures 4.5 and 4.8, overlain by the model grid. The actual monitored values of the final plume were also presented in Table 4.4.

It was anticipated that the unrealistic hydraulic boundary conditions implemented in stage two (due to data limitations) would influence the system's response to stresses imposed by the extraction operation. However, it was still hoped that the area of the grid where significant transport was occurring would be matched satisfactorily for the purposes of this study. The one change in contaminant transport boundary conditions for stage 3 was that the constant concentration cell representing the source was allowed to vary, modelling significant source material and contaminated soil removal. The flow and transport regimes for this stage of modelling were taken from previously listed assumptions and the calibration results of stages one and two, for transport and ground water flow respectively.

Figure 4.9 presents the modelled water-table configuration for the site under the hydraulic stresses produced by ten months of pump-and-treat system operation. Figure 4.10 shows the modelled solute plume after this same period of mass extraction through the pumping wells. The contaminant transport modelling results of Figure 4.10 can be qualitatively compared with the given distribution of Figure 4.8.

Residual calculations were performed in the same manner as in the previous two stages. Table 4.4 shows the results of these calculations for the contaminant concentration distributions of this stage. Since no given pumped regime data was available from the studies, residual calculations or qualitative comparisons of the water-table configuration were not undertaken.

As mentioned above, there were no observed pumped head distributions available for further calibration of the site's hydrogeology under these stresses. Some modification of this regime might have still been possible utilizing the plume's response to the pumping operation. Alternatively, calibration of the contaminant transport parameters might be possible, given the solute plume responses to the extraction system. However, simultaneous calibration of both these regimes for this stage would not result in a unique solution. Because of this data limitation, continued calibration was not attempted for this study, although it would be a highly recommended action for other remediation projects.

4.3.4 Discussion of Results

The calibration efforts described in section 4.3.2.1 produced a model of the hydrogeology and hydrostratigraphy that matched the observed water-table configuration satisfactorily. A quantitative analysis of residuals between modelling runs of this stage produced a minimized average value of 1.82 (1.19 in the transport zone) (Table 4.2). The model solution that resulted in this residual field also showed acceptable qualitative agreement with the observed water-table (Figures 4.2 and 4.3). Some concern has been raised over whether the boundary conditions overly constrained the model solution. Though this may have occurred, few other options were possible given the limited data available for this modelling effort.

A simple sensitivity analysis was performed on the effect of the modelled hydraulic conductivity zones within the aquifer. The qualitative difference between the resulting water-table of this simplified model (Figure 4.4) and that derived earlier using the heterogeneous conductivity (Figure 4.3) can be seen by comparing the two figures. An average residual of 3.00 (2.88 in the transport zone) was calculated for the water-table configuration of this simplified regime using the same method as in other ground water flow model stages. The increase in average residual for this simplified model indicates that variation of hydraulic conductivity may be one solution to the observed ground water regime. However, as discussed in section 2.6.4.3, this solution may not be unique in its close match to the flow field.

The ground water flow model of stage one was necessarily vague. Since no data was made available that defined this historical flow regime, simplifying assumptions described previously in this chapter were used to produce a flow field that agreed with the presented configuration of the contaminant plume. Obviously, a variety of different water-table configurations could have matched the contaminant plume end product equally well. The results of transport modelling for this stage were simply used as input of initial concentrations for the model grid of stage 3. Limited calibration of contaminant transport parameters was performed at this stage and the results viewed with some caution. However, the final calibrated values' agreement with typical values, as listed in the database of Tables A.2 and A.3 (Appendix A), did engender some degree of confidence in the modelling effort.

The average log-residual between the modelled plume configuration and the observed data was 1.39. The high concentration gradients around the source may have caused some elevated residuals due to the inaccuracy involved in locating the monitoring points within the model grid. Also, the linear averaging method used to determine values between cells may have skewed results somewhat, though it may have been acceptable practice in lower concentration gradient areas.

The contaminant plume that resulted from modelling 10 months of pump-and-treat system operation did match the observed behaviour in quantitative analysis (Table 4.4) to a similar degree as the stage one modelling. However, the qualitative comparison (Figures 4.8 and 4.10) is less obvious. It is interesting to note that the model results showed less plume withdrawal response than was observed in the field. The necessary assumptions and simplifications used in numerical modelling usually result in overly optimistic predictions, due to the fact that many of them leave out processes that have a negative impact on the remediation process. There are several processes that could produce increased concentration reduction that were not modelled in this study: increased dilution through recharge, increased decay action (through chemical or biochemical processes), increased partitioning out of the aqueous phase, etc. These processes or simply inaccurate calibration of modelled processes could result in the disagreement between model results and field data seen here.

Further calibration of either the ground water flow model or the contaminant transport model might have been undertaken at the third stage. However, lack of support data for these calibrations would make have made this a questionable effort. Inverse modelling techniques require an iterative relationship between the field responses and the model modifications. Continuing calibration efforts without supporting the results through extended definition of the site would not necessarily result in a more accurate solution. Directed investigation of questionable parameters or dependent variable values would be necessary to increase confidence in the results as model resolution was increased. The same data limitation for continued calibration cited above constrains further modelling effort with expanded process modelling capabilities.

Section 2.6.4.2 discussed the possible causes of inaccurate results in numerical modelling of ground water flow and contaminant transport. These were: uncertainty in model input values for material parameters, initial values and regime geometry; inaccuracies in the modelled governing equations and boundary conditions; and errors in the numerical solution to the governing equations. It is apparent that each of these sources of error might apply to this modelling effort, in varying degrees. Data limitations for material properties in flow and transport, initial values for the dependent variables, and geometry of the aquifer probably resulted in more significant errors than those proceeding from inaccurate assumptions or boundary conditions, and numerical errors in this case.

4.4 SUMMARY

At the beginning of this chapter, a summary of pump-and-treat methodology, from site characterization through closure, was presented. Figure 4.1 illustrated the interactions and relationships between various components of the operation using an idealized flowchart. This flowchart was related to the case study experiences in pump-and-treat system operation as described in Chapter 3. A discussion of the interaction between this empirical methodology and the flowchart conceptualizing the scientific aspects of the regime (Figure 2.1) was then undertaken.

Definitions of the terminology for inference, prediction, and direct characterization were then presented, based on the usage within the current and previous chapters in this study. Using this terminology, a discussion of inverse numerical modelling as an inference method was given, particularly as used for ground water flow and contaminant transport parameter definition. This was followed by an extensive listing of available numerical and analytical models for processes occurring in these two areas, especially as relating to pump-and-treat system design and operation. The listing included modelling packages for: ground water flow, solute transport, multi-phase flow and transport, flow and transport parameter estimations, geostatistical packages (for parameter estimation or distribution), capture zone estimation or well configuration optimization, and hydro(geo)chemistry. A discussion of the present day capability in coupling the various processes in models followed this listing.

An example of inference and prediction application was then presented that attempted to couple ground water flow and contaminant transport processes in a numerical model constructed with limited available data. This model encompassed three stages in the history of a case: contaminant plume development, interim action response in the ground water regime, and plume response to pump-and-treat system operation.

This modelling example illustrated two very important points. First, it is possible to obtain a working model of the ground water regime and contamination processes for a given site with very limited data. Second, it is demonstrated that inverse or predictive modelling of a complex regime requires an iterative relationship between the characterization and system operation stages. The model produced by this effort cannot be positively identified as the correct configuration without continued agreement to field behaviour over a range of (natural or artificial) changes in the system stresses. This would not only eliminate alternative solutions that equally match the site's behaviour, but would also support the findings of the inference procedure with hard data. The extended investigation could be directed by the inference procedure in order to more efficiently use project resources.

Table 4.1: Commercial Models in Contaminant Transport

Model Name	Dimensions, Model Type	Capabilities	Authors/Developers
GROUND WATER FLOW MODELS			
FLONET	2D section, streamline based finite element	steady-state, heterogeneity, anisotropy, deformable grid, confined / unconfined aquifers	Waterloo HydroGeologic Software, Waterloo, Ontario Canada
FLOWCAD	2D, finite difference	transient flow, heterogeneity, anisotropy, confined / leaky-confined / unconfined aquifers	Waterloo HydroGeologic Software, Waterloo, Ontario Canada
FLOWPATH	2D, finite difference	steady state flow, heterogeneity, anisotropy, confined / leaky-confined / unconfined aquifers	Waterloo HydroGeologic Software, Waterloo, Ontario Canada
JDB2D/3D	2D / quasi-3D, finite difference	transient flow, confined / leaky-confined aquifers	Bredehoeft, J. D. (U.S. Geological Survey)
MICRO-FEM	multiple aquifers, finite element	transient flow, heterogeneity, anisotropy, confined / semi-confined / unconfined / stratified / leaky aquifers	Hemker, C. J., and van Elburg, H.
MODFE	2D area and axisymetric; finite element	transient flow, heterogeneity, confined/leaky-confined /unconfined aquifers	Cooley, R. L., and Torak, L. J. (U.S. Geological Survey)
MODFLOW	2D a[illegible] section, quasi- or fully-3D; finite difference	transient flow, heterogeneity, anisotropy, irregular geometry, layered semi-confined / unconfined aquifers	McDonald, M. G., and Harbaugh, A. W. (U.S. Geological Survey)
NETFLO	3D, equivalent series and parallel flow members (1D)	steady state flow, heterogeneity, discrete fractures capability	Ramarao, B. S. and Pahwa, S. B. (Intera Environmental Consultants)
PAT	analytical pathline and travel time model	infinite / semi-infinite, homogeneity, isotropy, confined aquifer (approximation of unconfined)	Kinzelbach, W., (University of Kassel) and Rausch, R. (University of Stuttgart)

Table 4.1: Commercial Models in Contaminant Transport

Model Name	Dimensions, Model Type	Capabilities	Authors/Developers
PLASM	2D, finite difference	transient flow, heterogeneity, anisotropy, unconfined / leaky-confined / confined aquifers	Prickett, T. A., and Lonnquist, C. G. (Illinois State Water Survey)
RADFLOW	radial flow, finite difference	transient flow, homogeneity, isotropy, confined / unconfined aquifers	Rathod, K. S., and Rushton, K. R. (University of Birmingham)
SEEP/W	2D area or section / 3D axisymmetric, finite element	transient flow, unsaturated. flow, confined / unconfined aquifers, heterogeneity, anisotropy	GEO-SLOPE International Ltd., Calgary, Alberta, Canada
CAPZONE	2D, analytical	homogeneity, isotropy, confined / leaky-confined / unconfined aquifers	Bair, E.S., Springer, A. E., and Roadcap, G. S. (Ohio State University)
GWFLOW	program package, analytical	transient flow, homogeneity, isotropy, confined / leaky-confined / 2 mutually leaky-confined / unconfined aquifers	van der Heijde, P. K. M. (IGWMC)
THWELLS	Well flow, analytical drawdown calc.	transient flow, homogeneity, isotropy, confined / leaky-confined / unconfined aquifers	van der Heijde, P. K. M. (IGWMC)
TWODAN	analytical	transient flow, heterogeneity, one or two strata, confined / unconfined aquifers,	Fitts, C. (University of Southern Maine)
UN-GW	program package	g.w. chemistry, pumping tests, well hydraulics, water level database, well logs, cross sections, 2D areal finite difference flow model (transient flow, heterogeneity, isotropy, confined aquifer, 2D areal finite difference flow model (transient, heterogeneity, unconfined / confined aquifer)	Karanjec, J. K., and Braticevic, D. (U.N. Department of Technology Co-operative for Development)
WHPA	2D (area) semi-analytical	steady state flow, homogeneity, Monte-Carlo analysis	Blanford, T. N., Huyakorn, P. S., and Wu, W-S. (HydroGeologic Inc.)

Table 4.1: Commercial Models in Contaminant Transport

Model Name	Dimensions, Model Type	Capabilities	Authors/Developers
FLOW AND TRANSPORT MODELS			
AGU-10	program package		Javendel, I., Doughty, C., and Tsang, C. F. (Lawrence Berkeley Laboratory, CA)
RESSQ	2D, semi-analytical	solute transport; homogeneity, isotropy, confined aquifers, uniform thickness, steady state flow field, advection / adsorption	
LTIRD	Radial, analytical	solute transport; radial flow field, homogeneity, isotropy, uniform thickness aquifer, zero background conc.	
ODAST	1D, analytical	solute transort; steady state flow, homogeneity, isotropy aquifer, uniform thickness, advection / dispersion / decay / adsorption, zero background conc.	
TDAST	2D, analytical	solute transport; steady state flow, homogeneity, isotropy, uniform thickness aquifer, advection / dispersion / decay / adsorption, zero background conc.	
RT	Radial	steady state radial flow, time series conc. to spatial distribution	
ASM	2D, finite difference	transient flow, heterogeneity, anisotropy, confined / leaky-confined / unconfined aquifers; transient transport in steady state flow field	Kinzelbach, W., (University of Kassel) and Rausch, R. (University of Stuttgart)
BEAVERSOFT	program package, 2D, analytical / numerical	steady and unsteady state flow, heterogeneity, flow through dams, advection / dispersion	Bear, J., (Technion) and Verruijt, A. (Delft Technical University)

Table 4.1: Commercial Models in Contaminant Transport

Model Name	Dimensions, Model Type	Capabilities	Authors/Developers
FTWORK	1D / 2D (area and secional) / 3D, finite difference	transient flow, heterogeneity, anisotropy, confined / unconfined aquifers; transient tranport, advection / dispersion / linear-nonlinear adsorption / first-order decay. Parameter estimation for flow (hydraulic conductivity & recharge)	Faust, C. R. *et al.* (GeoTrans, Inc.)
HST3D	3D, finite difference	flow / heat and solute transport, advection / dispersion / linear adsorption / first-order decay	Kipp, K. L. (U. S. Geological Survey)
MYGRT	2D (area or section), quasi-analytical	homogeneity, isotropy, horiz. steady state flow, advection / dispersion / linear adsorption / first-order decay,	
PRINCE	program package	1D/2D/3D analytical transport, 2D analytical flow, homogeneity, isotropy, steady state, advection / dispersion / linear adsorption / first-order decay	Waterloo HydroGeologic Software. Waterloo, Ontario Canada
RANDOM WALK	2D, analytical and finite difference	transient flow, heterogeneity, confined / leaky confined / unconfined aquifers, advection / dispersion / first-order decay-production / linear adsorption / zero-order production	Prickett, T. A. *et al.*. (Illinois State Water Survey)
RWH	1D/2D, particle-in-cell convection, random walk dispersion	homogeneity, isotropy, confined aquifer, steady state flow, advection / dispersion / linear adsorption / first-order decay	van der Heijde, P. K. M. (IGWMC)
SUTRA	2D area or section, hybrid finite element / integrated finite difference	saturated (area/section) / unsaturated (section) flow	Voss, C. I. (U. S. Geological Survey)

Table 4.1: Commercial Models in Contaminant Transport

Model Name	Dimensions, Model Type	Capabilities	Authors/Developers
SWICHA	2D (area or section) / axisymmetric / 3D, finite element	variable density flow and transport, transient flow. advection / dispersion / absorption / first-order decay	Lester, B. (GeoTrans, Inc.)
TRAFRAP-WT	2D, finite element	steady state flow, leaky-confined / unconfined aquifers, discrete-fracture or dual porosity fractured aquifers; advection / dispersion in fractures, diffusion in host media, first order decay	Huyakorn, P. S. *et al.* (HydroGeologic Inc. for IGWMC)
VERTPAK-1	program package	BAREN: radial, analytical, double porosity, transient flow, homogeneity, isotropy, fractured-confined aquifer. GIBMAC: plane strain / axisymmetric consolidation, semi-infinite, saturated, homogeneity, isotropy. GRINRH: axisymetric, analytical, transient flow, horizontal fracture. GRINRV: 2D, analytical, transient flow, vertical fracture. HART: thermoelastic deformation. LESTER: 1D, dual porosity, solute transport, advection / dispersion. STRELT: radial, transient flow, dual porosity, confined aquifer. TANG: planar fracture, solute transport, 1D advection / dispersion / first-order decay / adsorption along fracture, 1D diffusion / first-order decay / adsorption into matrix	Intera Environmental Consultants, Inc., Austin, TX)
WALTON 35	program package	35 analytical flow, solute, and heat transport models	Walton, W. C.

Table 4.1: Commercial Models in Contaminant Transport

Model Name	Dimensions, Model Type	Capabilities	Authors/Developers
SOLUTE TRANSPORT MODELS			
AT123D	1D/2D/3D, analytical	transient transport, homogeneity, anisotropy, uniform flow, dispersion / linear adsorption / first-order decay	Yeh, G. T. (Oak Ridge National Laboratory)
BIOPLUME II	2D, method of characteristics	coupled (superposition) hydrocarbon and oxygen transport, convection / dispersion / oxygen-limited biodegradation / first-order decay (reaeration and anaerobic biodegradation)	Rafai, H. S. *et al.* (Rice University)
CANVAS	2D, analytical / finite element	viral particle transport, saturated / unsaturated flow and transport, steady state flow, advection / dispersion / adsorption / inactivation / colloidal filtration	Park, N-S., Blanford, T. N., Wu, Y-S., and Huyakorn, P. S. (HydroGeologic Inc.)
CTRAN/W	2D, finite element (post SEEP/W)	transient and/or unsaturated transport in steady state flow fields, heterogeneity, anisotropy, advection / dispersion / diffusion(fn()) / linear adsorption / first-order decay, fractured rock transport simulation	GEO-SLOPE International Ltd., Calgary, Alberta, Canada
EPA/VHS	1D, analytical	screening level (max. conc.), steady state horizontal flow, conservative solute, homogeneity, isotropy, transverse dispersion	van der Heijde, P. K. M. (IGWMC)
FLOTRANS	2D (section), finite element (post FLONET)	transient transport, heterogeneity, anisotropy, confined / unconfined aquifers, advection / dispersion / linear adsorption / first-order decay	Waterloo HydroGeologic Software, Waterloo, Ontario Canada
MOC (USGS 2-D)	2D, finite difference	heterogeneity, anisotropy, confined aquifers, advection / dispersion / mixing-dilution / first-order decay / linear-Freundlich-Langmuir sorption / ion exchange (mono- and divalent)	Konikow, L. F., and Bredehoeft, J. D. (U. S. Geological Survey)

Table 4.1: Commercial Models in Contaminant Transport

Model Name	Dimensions, Model Type	Capabilities	Authors/Developers
MOCDENSE	2D (section), finite difference	advection / dispersion, one or two species, density driven flow	Sanford, W. E., and Konikow, L. F. (U. S. Geological Survey)
MT3D	3D, finite difference	advection / dispersion / sink-source mixing / linear or non-linear sorption / first-order decay, confined / unconfined / variably confined/unconfined aquifers	Zheng, C. (S. S. Papadopulos & Assoc.)
ONE-D	program package, 1D analytical	advection / dispersion / diffusion / linear adsorption / zero-order production / first-order decay	van Genuchten, M. Th., and Alves, W. J. (USDA Salinity Laboratory
PLUME	2D / 3D, semi-analytical	uniform flow, homogeneity, advection / dispersion / radioactive decay	van der Heijde, P. K. M. (IGWMC)
PLUME2D	analytical	homogeneity, confined aquifer, uniform horizontal flow, advection / dispersion / linear adsorption / first-order decay	van der Heijde, P. K. M. (IGWMC)
SOLUTE	program package	1D / 2D (area and section) / Radial / 3D, analytical, advection / dispersion / linear adsorption / first-order decay	Beljin, M. S. (IGWMC)
USGS-SOL	program package	1D/2D/3D, analytical, uniform flow, Gaussian concentration, advection / dispersion / linear adsorption / ion exchange / first-order decay	Wexler, E. J. (U. S. Geological Survey)
VIRALT	semi-analytical and numerical	saturated/unsaturated transport and fate of viruses, steady state and transient, advection / dispersion / adsorption / inactivation, transport	Park, N-S., Blanford, T. N., Wu, Y-S., and Huyakorn, P. S. (HydroGeologic Inc.)
UNSATURATED FLOW AND/OR TRANSPORT MODELS			
CHEMFLO	1D, finite difference	screening level, advection / dispersion / first-order decay (liquid and solid phase) / zero-order production-decay / linear adsorption	Nofziger, D. L., Rajender, K., Nayudu, S. K., and Su, P. Y. (Oklahoma State University)

Table 4.1: Commercial Models in Contaminant Transport

Model Name	Dimensions, Model Type	Capabilities	Authors/Developers
HYDRUS	1D, finite element	transient flow and solute transport in variably saturated media, diffusion / dispersion / linear-nonlinear adsorption / first-order decay	Kool, J. B. (HydroGeologic) and van Genuchten, M. Th. (U. S. Salinity Laboratory)
INFIL	1D, analytical	ponded infiltration, homogeneity, unsaturated media	Vauclin, M. (France), and El-Kadi, A. I. (IGWMC)
PESTAN	1D	vertical transport of organics, homogeneity, dispersion / first-order degradation, linear adsorption	Enfield, C. (USEPA/RSKERL)
PESTRUN	analytical	sorption / microbial-chemical-photo-degradation / volatilization, erosion and runoff mass loss	McCall, E. C., and Lane, D. D. (Kansas Water Resources Research Institute)
PRZM2	1D, **PRZM**: finite difference **VADOFT**: finite element	PRZM: fate and transport of solute and two daughter species, volatilization / vapour transport / biotransformation. VADOFT: unsaturated flow	Mullins, J. A. *et al.* (EPA Center for Exposure Assessment Modeling)
RITZ	analytical	screening level, unsaturated flow and transport, advection / volatilization / degradation / linear adsorption / first-order decay, uniform water flow field	Nofziger, D. L. (Oklahoma State University), Williams, J. R. and Short, T. E. (U.S. EPA)
SUMATRA-1	1D, Hermitian finite element	flow and transport of oil in saturated and moderately unsaturated soil, heterogeneity, linear adsorption / zero- and first-order decay	van Genuchten, M. Th. (Princeton University)
SWACROP	1D, finite difference	transient unsaturated flow	Wesseling, J. G., Kabat, P., van den Brock, B. J., and Feddes, R. A.
SWMS_2D	2D (area or section) or 3D (radial).	water and solute transport in variably saturated media, irregular boundaries, heterogeneity, anisotropy, advection / dispersion / linear adsorption / zero-order production / first-order decay	Simunek, J., Vogel, T., and van Genuchten, M. Th. (U.S. Salinity Laboratory, Riverside, CA)

Table 4.1: Commercial Models in Contaminant Transport

Model Name	Dimensions, Model Type	Capabilities	Authors/Developers
VLEACH	1D, finite difference	steady state flow, homogeneity, advection / volatilization / adsorption / gas diffusion, no dispersion or degradation/production	Turin, J., Carlsson, F., Sukop, M., and Lawson, P. (CH2M Hill, Redding CA)
VS2D/T	2D (section or cylinder), finite difference	variably saturated flow and transport, non-linear storage / infiltration / EVT / first-order decay / Freundlich or Langmuir adsorption / ion exchange	Healy, R. W. (U. S. Geological Survey)
MULTI-PHASE FLOW AND TRANSPORT			
ARMOS	2D, finite element	water flow only / oil flow with steady state water / coupled oil-water flow, heterogeneity, anisotropy, unconfined aquifers	Environmental Systems and Technologies, Inc.
MOTRANS	2D (section and radial), finite element	water flow only / oil flow with steady state water / coupled oil-water flow / coupled air-oil-water flow, advection / dispersion / equilibrium partitioning or first-order non-equilibrium mass transfer, five partitionable species	Environmental Systems and Technologies, Inc.
SPILLCAD	graphical database	monitoring well and boring database, spill volume assessment, free product and dissolved plume migration assessment,	Environmental Systems and Technologies, Inc.
SPILLTRANS	2D(areal), numerical	solute transport and multi-phase flow, LNAPL and up to 5 dissolved species, advection / dispersion / adsorption / first-order decay and volatilization, heterogeneity	Environmental Systems and Technologies, Inc.
SWANFLOW	3D, finite difference	water and NAPL flow, saturated / unsaturated, no air flow, no partitioning	Faust, C. R., and Rumbaugh, J. O. (GeoTrans, Inc.)

Table 4.1: Commercial Models in Contaminant Transport

Model Name	Dimensions, Model Type	Capabilities	Authors/Developers
VAPOUR TRANSPORT PACKAGES			
AIRFLOW/SVE	radial-symmetric, numerical	transient vapour transport, multiple species, heterogeneity, kinetic and equilibrium three-phase partitioning	Waterloo Hydrogeologic Software, Waterloo, Ontario, Canada
HYPERVENTILATE	analytical	screening level, identify data requirements, evaluate soil permeability test results, minimize extraction wells, design removal rates	Johnson, P. C., and Stabenau, A. E. (Shell Development, Corporation)
VENTING	radial flow, finite difference	partitioning between air / water / NAPL / soil, equilibrium with efficiency factor	Environmental Systems and Technologies, Inc.
PARAMETER ESTIMATION - SATURATED FLOW			
AQUIX-4S	forward and inverse (least squares) model, flow and slug test data	transmissivity / storgae coefficient / leakage factor / anisotropy / specific yield	EnviroTools Ltd., Evergreen, Colorado
MODFLOWP	non-linear regression, weighted least squares, model input calibration, (after MODFLOW)	transmissivity / storage coefficient / hydraulic conductivity / specific yield, vertical leakance / vertical and horizontal anisotropy / hydraulic conductance	Hill, M. C. (U. S. Geological Survey)
ONESTEP	1D, finite element, outflow experiments	estimates five unknown parameters	Kool, J. B., Parker, J. C., and van Genuchten, M. Th. (Virginia Polytechnical Institute)
OPTP/PTEST	non-linear drawdown or single-variable constrained nonlinear, step-drawdown test data	optimal discharge calculation	Paudyal, G. N., and Gupta, A. D. (Asian Institute of Technology)
PUMPTEST	program package, pump test data	DISTANCE: transmissivity and storage coef., RECOVERY: transmissivity, UNITS: conversion	Beljin, M. S. (IGWMC)

Table 4.1: Commercial Models in Contaminant Transport

Model Name	Dimensions, Model Type	Capabilities	Authors/Developers
SLUGC	manual curve fitting, slug tests data	hydraulic conductivity	van der Heijde, P. K. M. (IGWMC)
SLUGT	analytical, slug test data	hydraulic conductivity	Mills, A. C. (Woodward-Clyde Consultants)
TGUESS	estimates from specific capacity	transmissivity	Bradbury, K. R., and Rothschild, E. R. (Wisconsin Geological Survey)
THCVFIT	radial flow, pumping test data	non-steady state, transmissivity, storage coefficients	van der Heijde, P. K. M. (IGWMC)
THEISFIT	least squares, aquifer test data	transmissivity, storage coefficients, homogeneity, anisotropy, nonleaky confined aquifers	McElwee, C. D. (Kansas Geological Survey)
TSSLEAK	least squares, pump test data	transmissivity, storage coefficients, leakage coefficients, aquitard permeability	McElwee, C. D. (Kansas Geological Survey)
VARQ	least squares, pump test data	aquifer parameters	Butt, M. A., and McElwee, C. D. (Kansas Geological Survey)
PARAMETER ESTIMATION - TRANSPORT			
CATTI	1D or 2D uniform or radial flow, non-linear least squares, tracer test data	breakthrough curves, aquifer parameters (longitudinal and transverse dispersivity), homogeneity, one or two layers	Sauty, J. P. (BRGM), and Kinzelbach, W. (University of Kassel)
CFITIM	1D transport models, miscible displacement data	steady state uniform flow field, advection / dispersion, equilibrium / non-equilibrium transport parameters (five), dispersion coefficient / distribution coefficient (linear or user-input)	van Genuchten, M. Th. (USDA Salinity Laboratory)

Table 4.1: Commercial Models in Contaminant Transport

Model Name	Dimensions, Model Type	Capabilities	Authors/Developers
CXTFIT	1D transport models, nonlinear least squares,	advection / dispersion / linear adsorption / zero-order production / first-order decay, non-equilibrium transport, stochastic flux distribution	Parker, J. C., and van Genuchten, M. Th. (Virginia Polytechnical Institute)
WELL	tracer test data	longitudinal dispersion (no transverse), effective porosity	Gelhar, L. W. (Massachusetts Institute of Technology)
PARAMETER ESTIMATION - UNSATURATED FLOW			
FP	experimental data	retention function parameters (imbibition etc.)	Su, C., and Brooks, R. H. (Oregon State University)
RETC	analytical, forward and inverse, soil water retention data	soil water retention curves, hydraulic conductivity curves, soil water diffusivity	van Genuchten, M. Th. *et al.* (USDA Salinity Laboratory, Riverside, CA)
SOHYP	analytical, nonlinear least squares	unsaturated hydraulic conductivity (closed form) function	van Genuchten, M. Th. (Princeton University)
SOIL	nonlinear least squares	soil water characeristic function, unsaturated hydraulic conductivity function	El-Kadi, A. I. (IGWMC)
SOILPROP	soil hydraulic properties and their uncertainty from particle size distributions	soil water content versus capillary pressure, and relative permeability	Environmental Systems and Technologies (Blacksburg, VA)
GEOSTATISTICAL PACKAGES			
COVAR	2D, normal or lognormal	generation of autocorrelated parameters	Williams, S. A., and El-Kadi, A. I. (IGWMC)
GEO-EAS	2D	statistical analysis of spatial data	Englund, E. (U. S. EPA), and Sparks, A. (Computer Sciences Corp.)

Table 4.1: Commercial Models in Contaminant Transport

Model Name	Dimensions, Model Type	Capabilities	Authors/Developers
GEOPACK	2D	statistical and mathematical analysis of spatial data	Yates, S. R. (USDA/ARS), and Yates, M. V. (University of California)
MAP	program package	PLUME: 2D analytical transport in uniform flow field. MEMO: optimization of monitoring well locations. COPRO: probabilities of downstream contamination.	Golder Associates
POST-MODFLOW MODELS			
ZONEBUDGET	3D (post-MODFLOW)	Water budget model	Harbaugh, A. W. (U. S. Geological Survey)
MODPATH	3D (post-MODFLOW), semi-analytical particle tracking	Capture zone model	Pollock, D. W. (U. S. Geological Survey)
PATH3D	2D / 3D, particle tracking finite difference (post-MODFLOW)	Capture zone model, steady state / transient flow	Zheng, C. (S. S. Papadopulos & Assoc.)
MODMAN	(post MODFLOW)	location of pumping/injection wells, extraction/injection rate, maximization or minimization	Greenwald, R. M. (GeoTrans, Inc.)
HYDRO(GEO)CHEMISTRY MODELS			
BALANCE	mass-balance reactions	geochemical reactions included, inversely matched reactions to observed conc. changes along flow path	Parkhurst, D. L., *et al.* (U. S. Geological Survey)
DSALT-2	mass-balance reactions	geochemical reactions included, complete mixing or breakthrough (salinization or desalinization), ESR, SAR, ion-exchange	Dudas, M. J., and Abley, M. (University of Alberta)

Table 4.1: Commercial Models in Contaminant Transport

Model Name	**Dimensions, Model Type**	**Capabilities**	**Authors/Developers**
MINTEQA2	(geo) chemical equilibrium	speciation, dilute aqueous systems, dissolved / adsorbed / solid phases / gas phases, chemical database	Allison, J. D., Brown, D. S. and Novo-Gradac, K. J. (U. S. EPA)
NETPATH	mass-balance reactions	geochemical reactions included, inversely calculates mixing of two sources based on conc. changes along flow path	Plummer, L. N., Prestemon, E. C., and Parkhurst, D. L. (U. S. Geological Survey)
PHREEQE	(geo)chemical reactions	non-equilibrium effects, ion-pairing, pH, redox potential, mass transfer, multiple phase solution concentration	Parkhurst, D. L., Thorstenson, D. C., and Plummer, L. N. (U. S. Geological Survey)
PHRQPITZ	(geo)chemical reactions	non-equilibrium effects, osmotic coefficients, water activity, mineral saturation indices, mean activity coefficients, total activity coefficients, pH, individual-ion activities, individual-ion acitivy coefficients	Plummer, L. N., *et al.* (U. S. Geological Survey)
SOLMINEQ.88	chemical equilibrium	speciation, ion pairing, ion activity products, percent free ions, free ion activities, disequilibria indices	Alberta Research Council
WATEQ4F	(geo) chemical equilibrium	thermodynamic speciation of inorganics (major and important minor) and complex species, pH, redox potential, solid and gaseous reactions	Ball, J. W., and Nordstrom, D. K. (U. S. Geological Survey)

Table 4.2: MODFLOW Model - Stage 2 Residual Calculations

Monitoring Well No.	X Coord.	Y Coord.	Modelled Value	Observed Value	Residual
	(ft)	(ft)	(ft)	(ft)	(ft)
1	1800	5300	91.67	89.5	2.17
2	1475	5200	88.94	86.7	2.24
3	1850	5150	89.32	87.4	1.92
4	2025	5150	89.33	87.8	1.53
5	2450	5100	87.60	81.2	6.40
6	1750	5000	87.71	85.9	1.81
7	1925	4925	86.80	85.9	0.90
8	1500	4875	86.68	85.7	0.98
9	1800	4850	86.46	85.3	1.16
10	1675	4750	86.22	85.6	0.62
11	2050	4450	84.55	84.1	0.45
12	300	4400	85.45	85.4	0.05
13	1300	4425	85.34	85.1	0.24
14	1550	4400	85.10	84.7	0.40
15	2450	4350	83.06	83.9	0.84
16	3900	4400	92.15	92.3	0.15
17	3200	4150	79.85	78.4	1.45
18	2500	3650	82.85	81.9	0.95
19	0	3500	83.58	83.3	0.28
20	1250	3500	83.63	83.1	0.53
21	1200	3200	82.71	83.0	0.29
22	200	2800	81.79	82.7	0.91
23	500	2900	81.29	82.8	1.51
24	600	2700	80.00	78.5	1.50
25	3600	2600	88.57	81.9	6.67
26	400	2000	84.83	82.4	2.43
27	600	2000	84.73	82.1	2.63
28	1700	1900	87.54	84.3	3.24
29	3600	1400	92.45	83.4	9.05
30	1350	1300	90.33	87.9	2.43
31	2250	1300	91.64	86.2	5.44
32	400	800	92.03	86.8	5.23
33	1200	600	93.69	92.6	1.09
34	2200	800	93.67	92.1	1.57
35	1950	5400	94.54	96.5	1.96
36	1975	5250	91.26	91.2	0.06

Average Residual:	1.82
Avg. in Transport Zone:	1.19

Residual Formula: ABS(ModelValue-ObsValue)

Table 4.3: MT3D Model - Stage 1 Residual Calculations

Monitoring Well No.	X Coord.	Y Coord.	Modelled Value	Observed Value	Log Residual
	(ft)	(ft)	(ppb)	(ppb)	
1	1800	5300	6.49E-03	200	2.30
2	1475	5200	1.38E-07	300	2.48
3	1850	5150	2.93E+01	12000	4.08
4	2025	5150	2.52E-03	3800	3.58
5	2450	5100	2.85E-11	0	0.00
6	1750	5000	2.10E+05	7100	5.31
7	1925	4925	5.19E+03	2500	3.43
8	1500	4875	1.42E+02	540	2.60
9	1800	4850	1.49E+06	27000	6.17
10	1675	4750	7.65E+04	25000	4.71
11	2050	4450	-8.05E-04	35	1.53
12	300	4400	6.58E-16	0	0.00
13	1300	4425	1.05E-03	0	0.00
14	1550	4400	4.71E+00	16	1.01
15	2450	4350	8.25E-18	0	0.00
16	3900	4400	-2.43E-25	0	0.00
17	3200	4150	4.19E-24	0	0.00
18	2500	3650	-6.84E-27	0	0.00
19	0	3500	-7.61E-20	0	0.00
20	1250	3500	1.71E-09	36	1.54
24	600	2700	6.45E-15	0	0.00
25	3600	2600	-8.61E-35	0	0.00
29	3600	1400	0.00E+00	0	0.00
30	1350	1300	-2.22E-29	0	0.00
34	2200	800	6.18E-43	0	0.00
35	1950	5400	6.86E-03	0	0.00
36	1975	5250	5.69E-02	1000	3.00
37	2300	4975	2.07E-10	225	2.35
38	1200	4700	5.45E-03	1	-2.26
39	1825	4250	4.77E+01	1500	3.16
40	1900	4100	3.40E-01	240	2.38
41	2000	4050	1.38E-04	45	1.64
42	900	4050	2.65E-10	0	0.00
43	1100	4050	8.82E-08	2	0.00
44	1100	3950	2.50E-08	7	0.78
45	1150	3750	3.35E-09	50	1.69
46	1750	3650	3.45E-05	7	0.78
47	1925	3600	-1.15E-07	6	0.70
48	1150	3400	1.09E-10	18	1.23

Average Log Residual:	1.39

Log Residual Formula: LOG10(ABS(1+ModelValue-ObsValue))

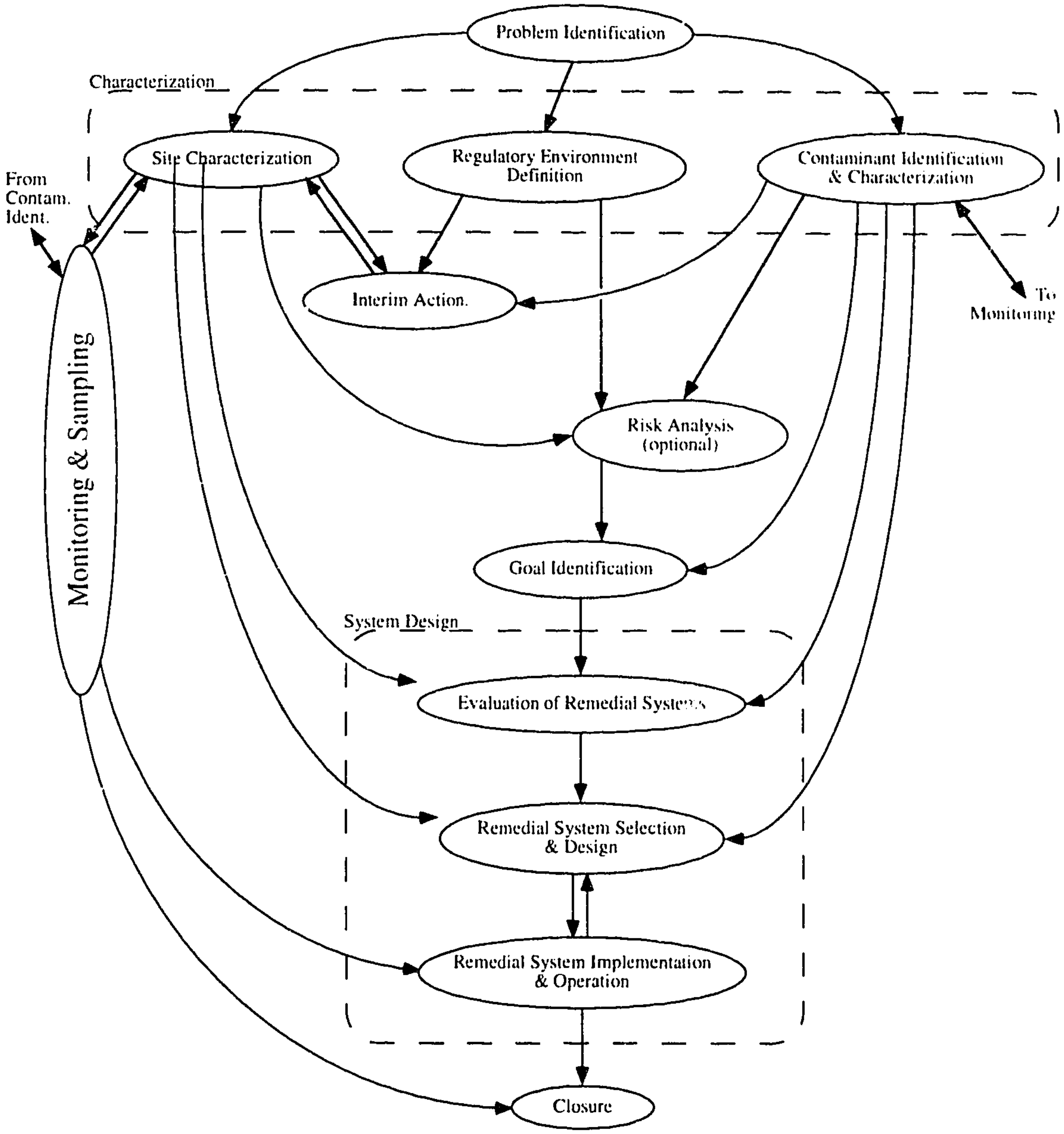
Table 4.4: MT3D Model - Stage 3 Residual Calculations

Monitoring Well No.	X Coord.	Y Coord.	Modelled Value	Observed Value	Log Residual
	(ft)	(ft)	(ppb)	(ppb)	
1	1800	5300	9.67E-01	13	1.04
2	1475	5200	-1.03E-05	1	-4.99
3	1850	5150	2.20E+01	2200	3.34
4	2025	5150	2.26E-01	141	2.15
6	1750	5000	6.03E-05	1800	3.26
7	1925	4925	2.83E+01	5000	3.70
8	1500	4875	8.85E+01	1600	3.18
9	1800	4850	1.31E-03	3900	3.59
10	1675	4750	5.40E-06	7700	3.89
11	2050	4450	1.35E+00	12	0.98
14	1550	4400	7.72E-14	0	0.00
15	2450	4350	2.08E+02	0	2.32
16	3900	4400	5.25E-02	0	0.02
17	3200	4150	1.12E+00	0	0.33
36	1975	5250	1.25E+00	70	1.83
37	2300	4975	-1.72E+01	1	1.23
38	1200	4700	4.08E-05	0	0.00
46	1750	3650	6.42E-17	3	0.30
49	1600	4800	4.87E-06	122	2.08
50	1450	4700	3.61E-12	0	0.00
51	2300	4675	1.02E+04	1	4.01
52	1100	4600	9.92E-04	0	0.00
53	1750	4600	4.27E-03	1000	3.00
54	1825	4625	1.01E-01	20	1.28
55	2200	4575	1.03E+04	3	4.01
56	2125	4500	4.89E+01	15	1.54
57	1050	4425	3.43E-03	0	0.00
58	1850	4400	7.84E-05	66	1.81
59	1975	4400	9.83E-03	0	0.00
60	1725	4100	8.83E-15	0	0.00
61	1825	3900	1.71E-13	0	0.00

Average Residual:	1.42

Log Residual Formula: LOG10(ABS(1+ModelValue-ObsValue))

Figure 4.1: Flowchart of Remediation System Evaluation, Design, and Operation



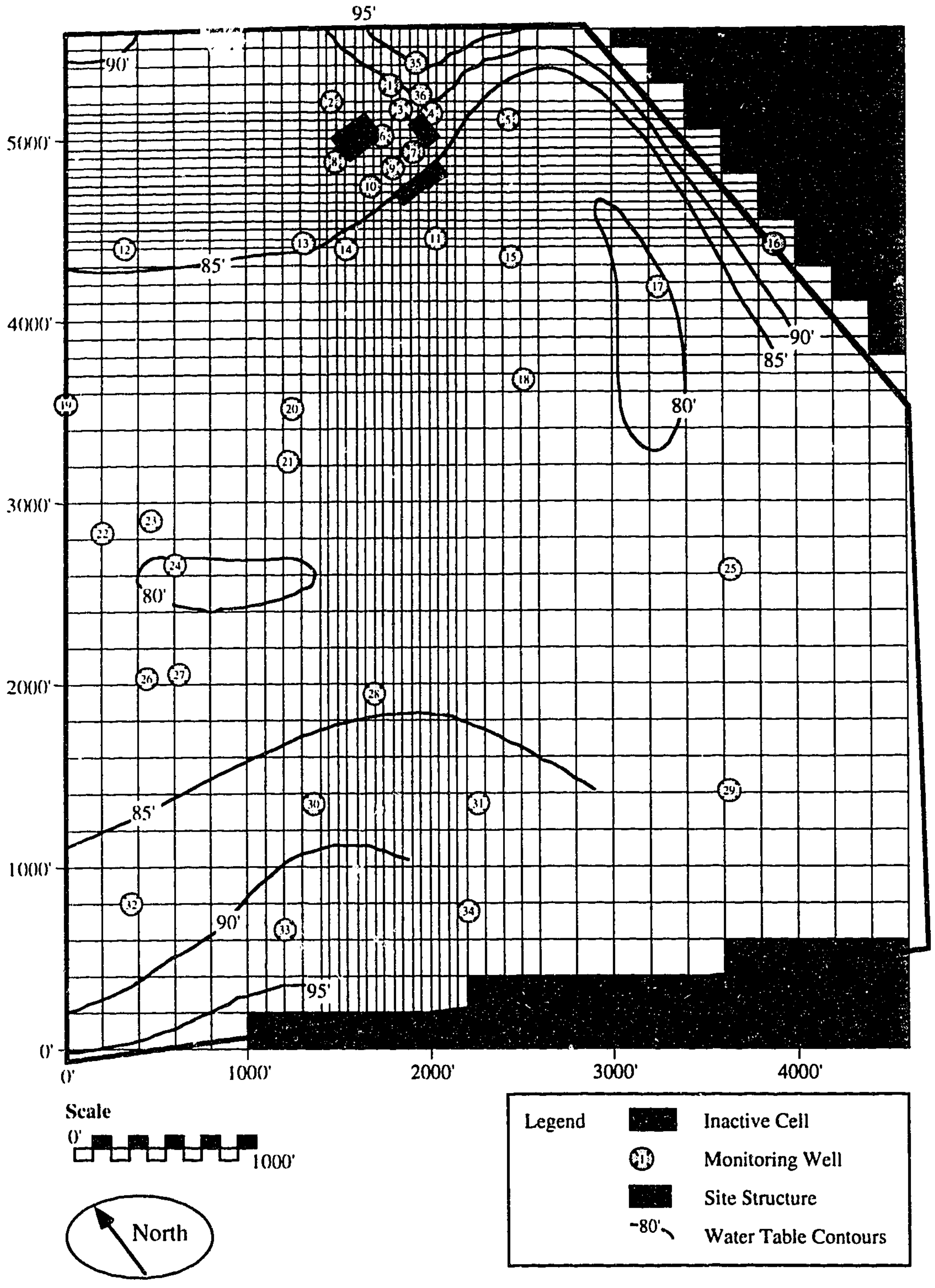


Figure 4.2: Site with Grid Layout and Measured Water-Table

Modified from Roux and Althoff (1980)

Figure 4.3: Modelled Stage 2 Water-Table

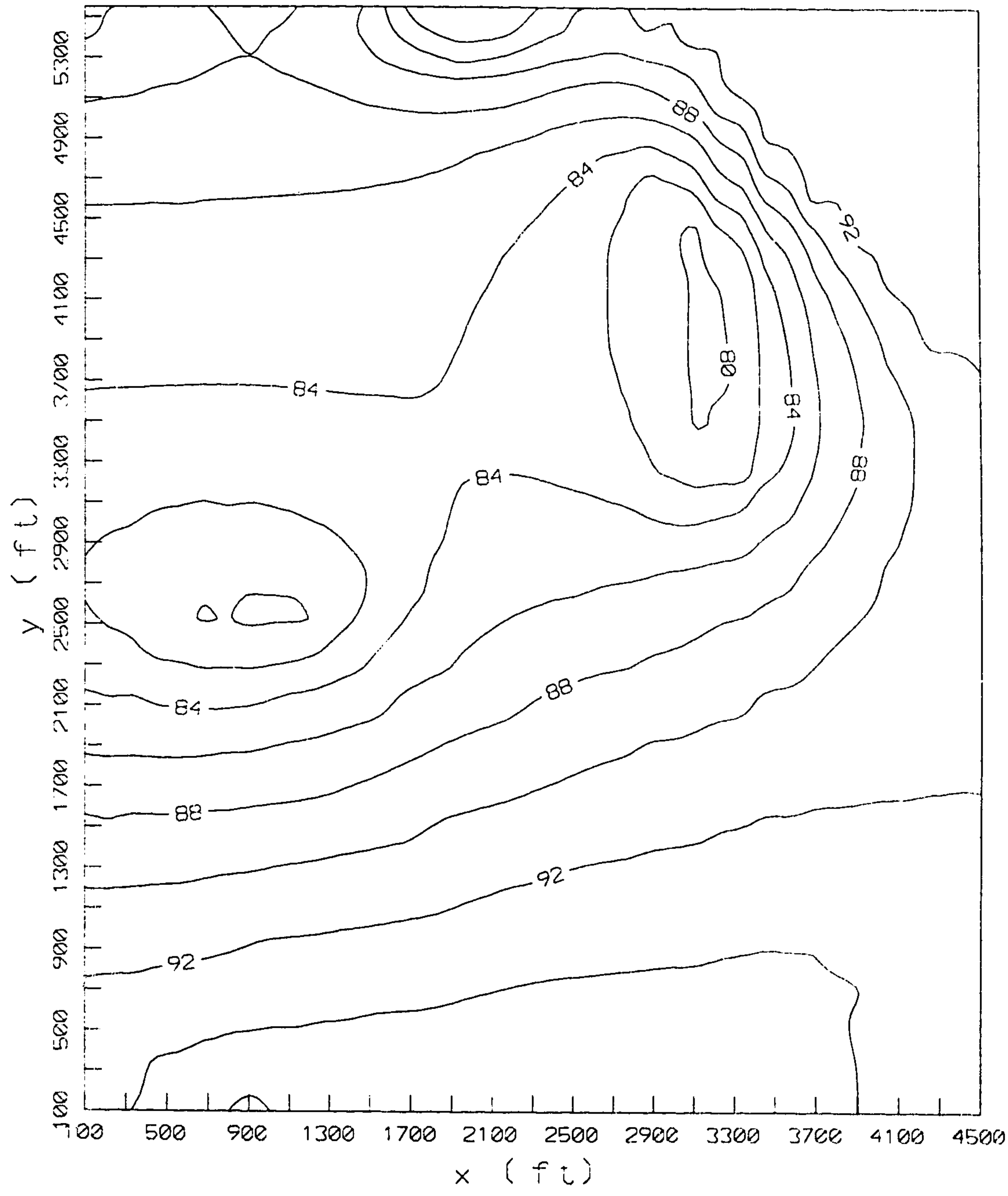
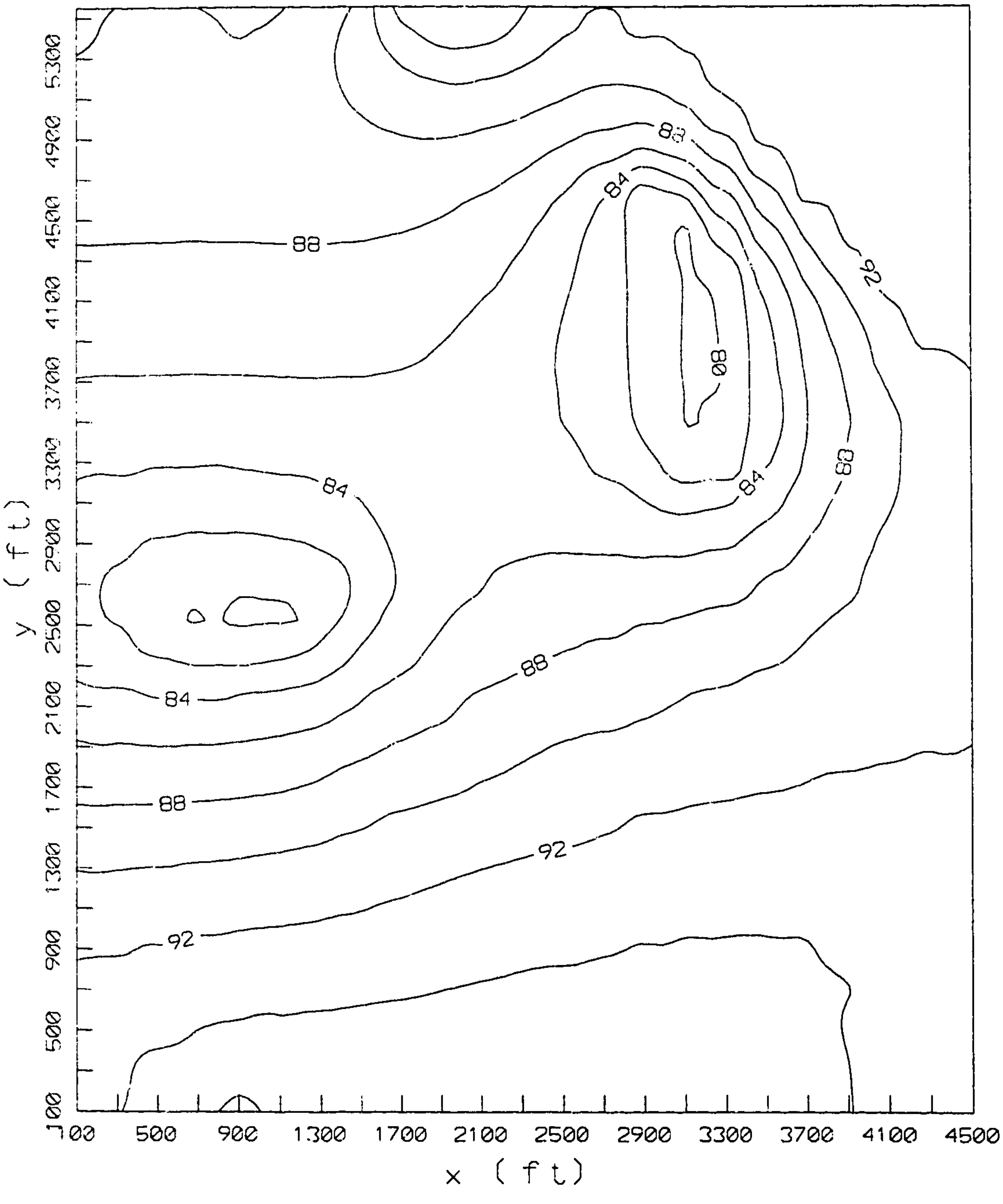


Figure 4.4: Modelled Stage 2 Water Table



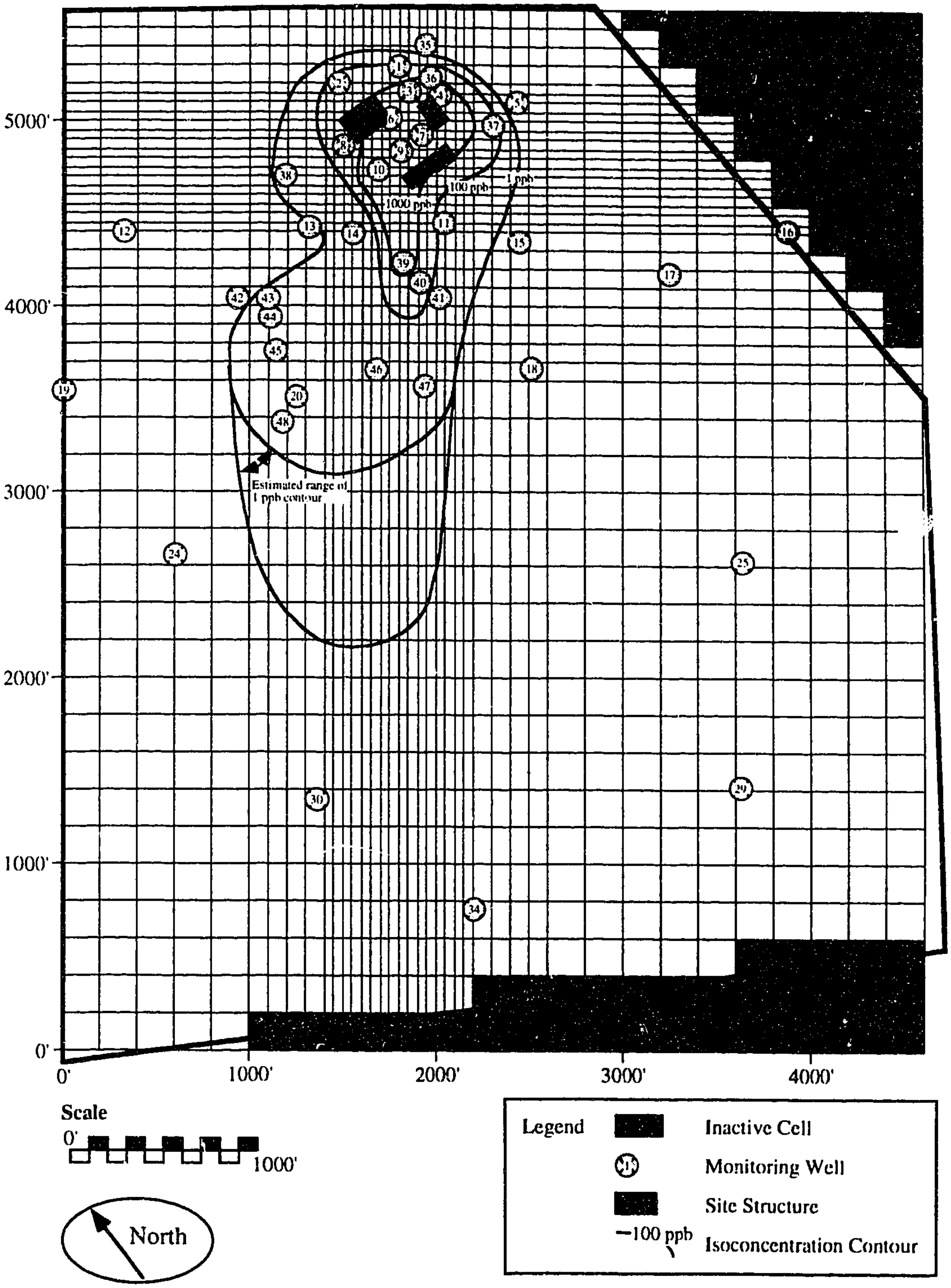


Figure 4.5: Site with Grid Layout and Measured Stage 1 Contaminant Plume

Modified from Althoff et al. (1981)

Figure 4.6: Modelled Stage 1 Contaminent Plume

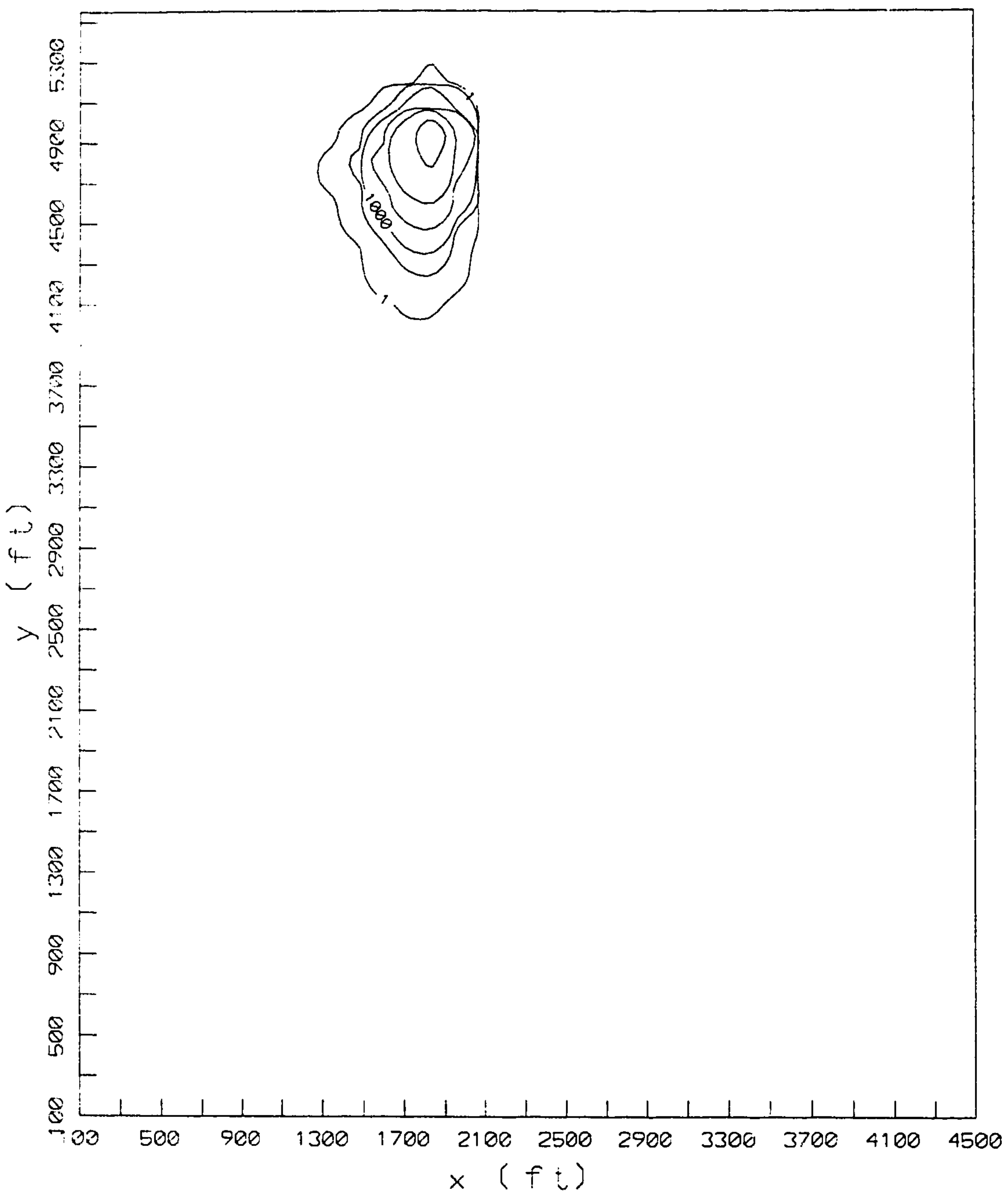
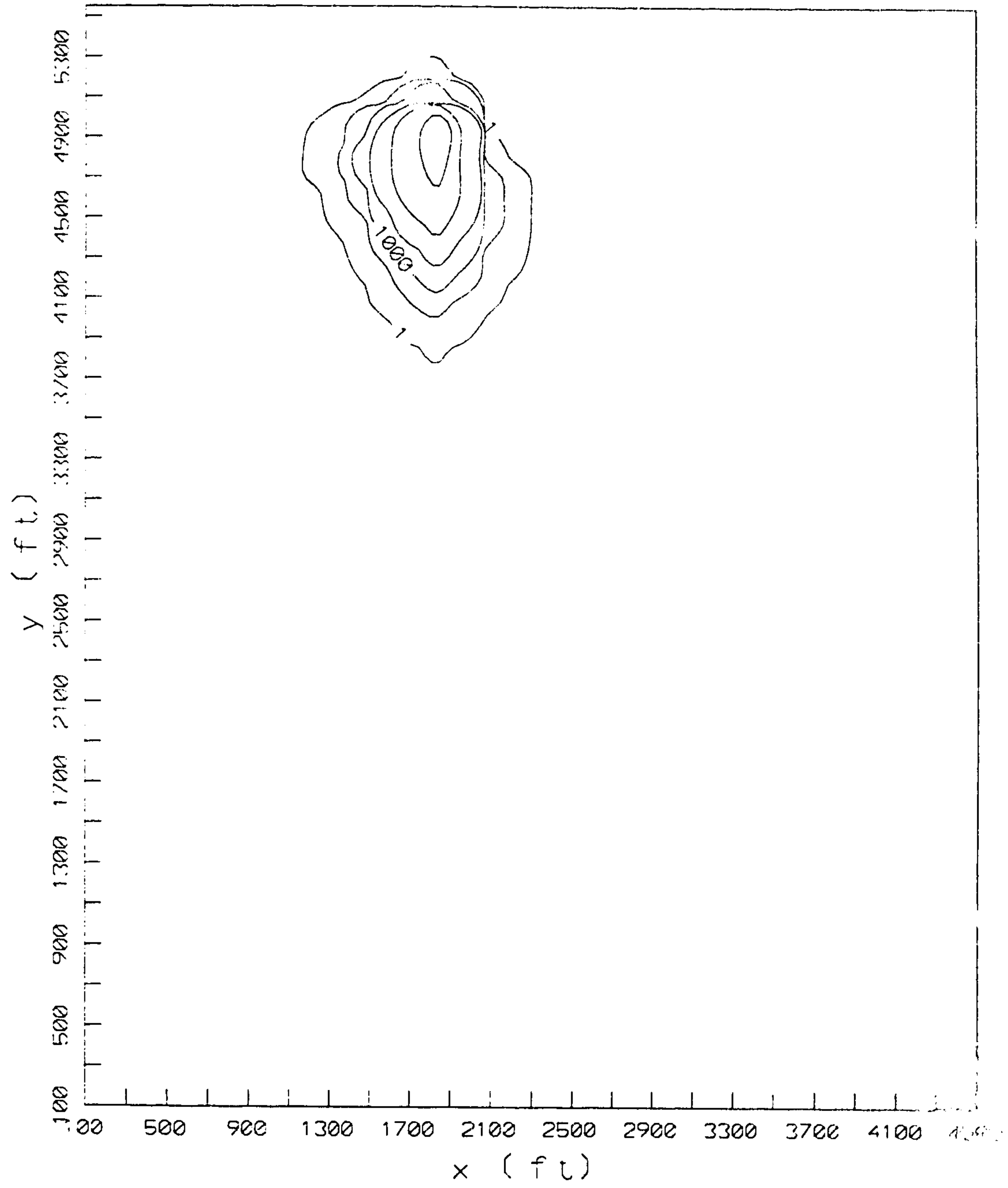


Figure 4.7: Modelled Stage 1 Contaminant Plume



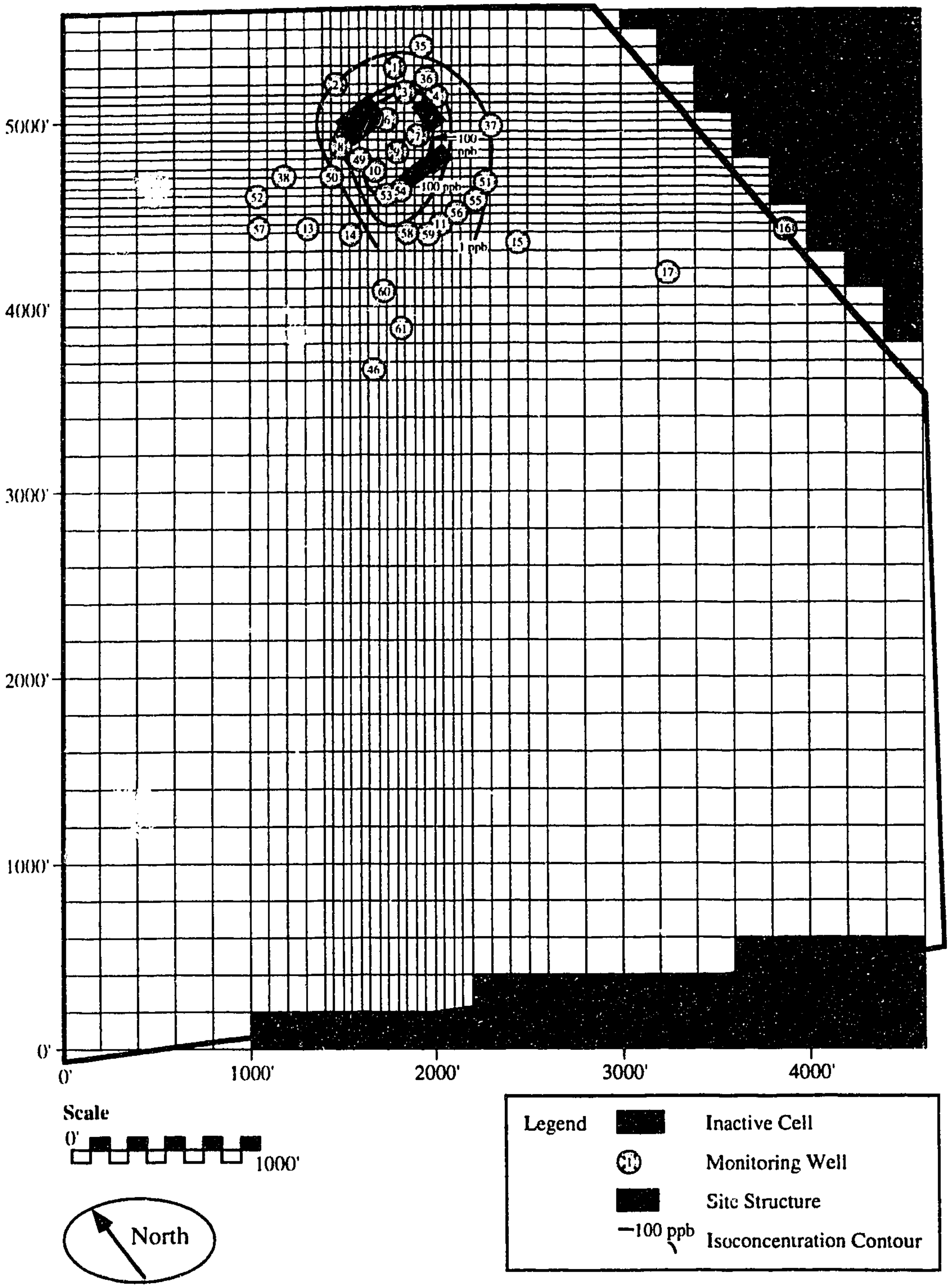


Figure 4.8: Site with Grid Layout and Measured Stage 3 Contaminant Plume
Modified from Althoff et al. (1981)

Figure 4.9: Modelled Stage 3 Water-Table

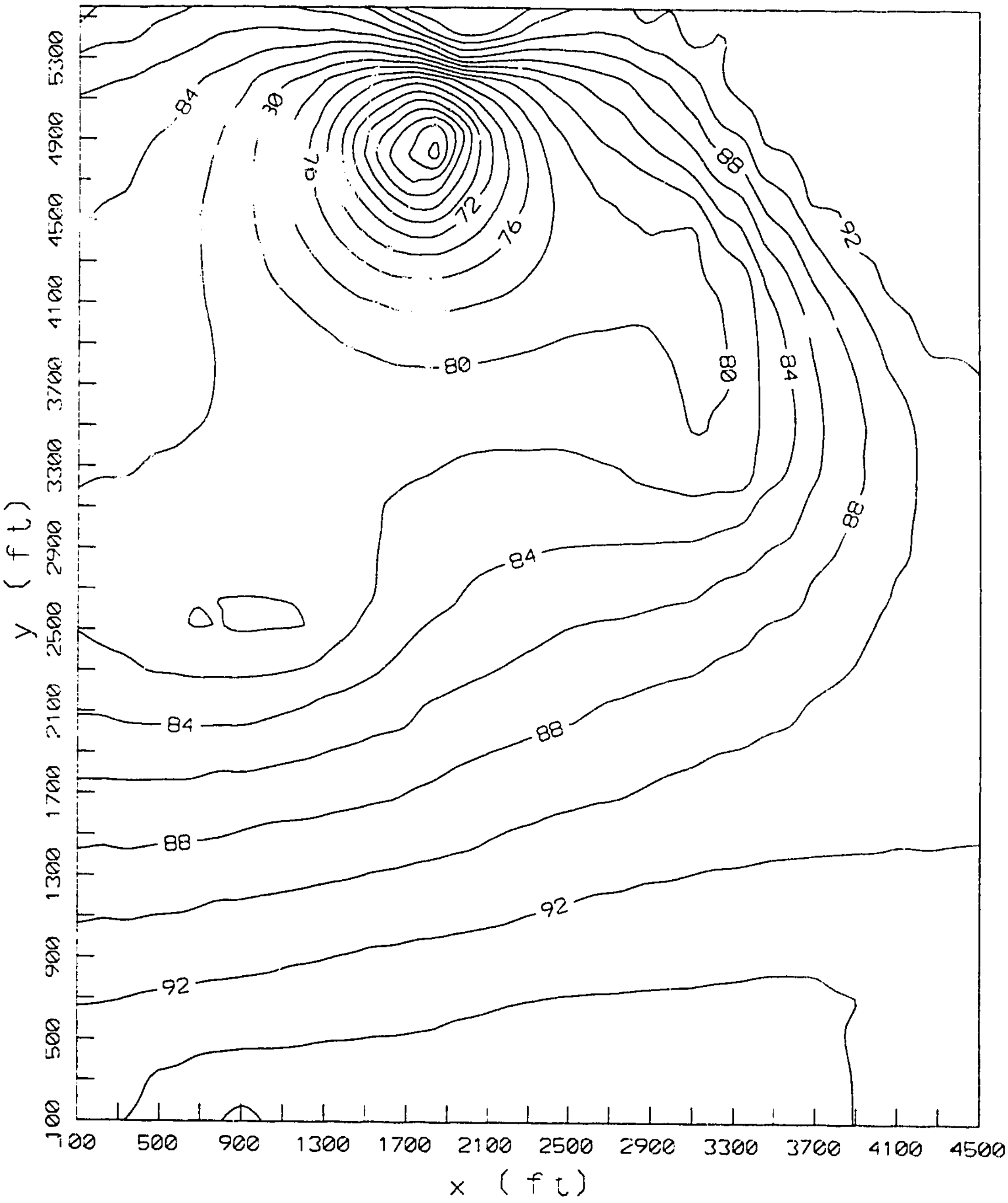
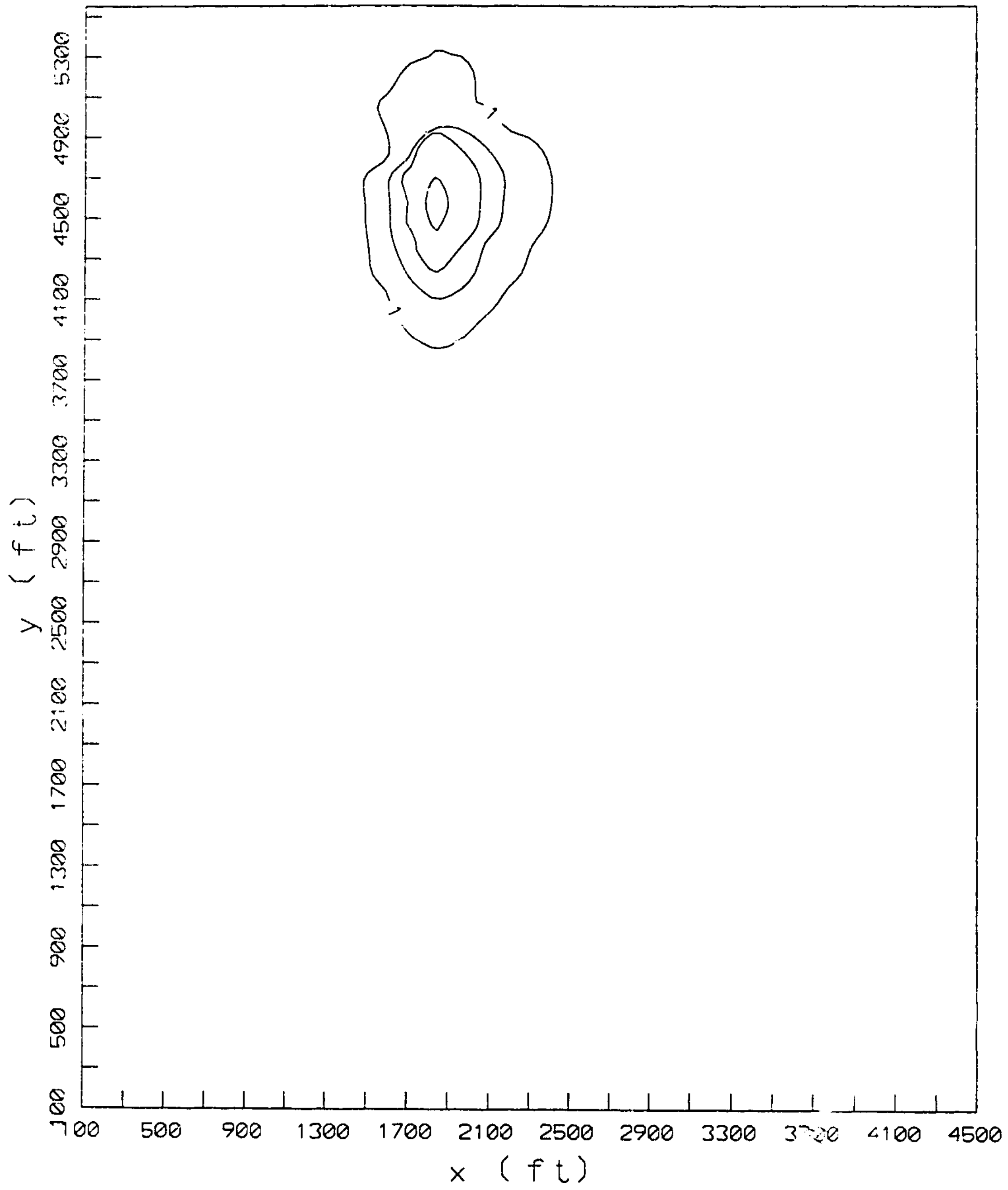


Figure 4.10: Modelled Stage 3 Contaminant Plume



CHAPTER 5: SUMMARY AND CONCLUSIONS

5.1 SUMMARY

Understanding of the properties and processes in the contaminant transport regime has greatly increased over the past few decades. However, this has failed to result in more consistently successful designs for pump-and-treat remediation systems. The uncertainty inherent in such projects has been met by few design methodologies that accept this reality and accommodate it in an efficient manner. A brief overview of the state of knowledge in the contaminant transport regime was introduced in a flowchart in Chapter 2 (Figure 2.1). This flowchart illustrated the major elements of the field, their interrelationships and interactions. Following this, a review of several case histories of pump-and-treat experience showed typical ranges of field conditions and responses. A linking of the scientific knowledge and empirical database provided by these two discussions led to an illustrative example of site characterization for pump-and-treat system design through inference and predictive modelling.

Arising from the study briefly described above came a conclusion that uncertainty is ubiquitous in subsurface remediation problems. Acceptance of the low probability of complete characterization in the design phase was therefore suggested, leaving the problem of how to manage this scenario. A methodology is required that accepts uncertainty and responds to it in a structured manner; anticipating possible deviations from previous site conceptualizations and accommodating their demands into the project design. The following discussion will provide a summary of the previous chapters' findings in more detail. A conclusion regarding the aforementioned methodology will then be introduced, followed by recommendations for further study.

5.1.1 Complexity in the Contaminant Transport Regime

Various elements of the contaminant transport regime were illustrated in the flowchart of Figure 2.1 and described in the text of Chapter 2, along with their interactions and influences. These components were categorized as solid matrix, fluid, or contaminant related properties or processes. The variability of these elements and their specific characteristics were then presented in more detail. General determination methods for these properties and processes were also introduced.

Site investigation, interim action, sampling and monitoring were listed as contributors to a remediation investigation process. The effect of physical and chemical processes on the worth of monitoring data was also considered. Inference and prediction methods were discussed in detail and conceptualized as boundary value problems. The effects of input data uncertainty on modelling results was discussed. Inference was suggested as a relatively efficient method of reducing model inaccuracies through calibration. Predictive modelling was then presented as an optimization technique for pump-and-treat system design. Additional difficulties that might appear upon full-scale operation of a system were briefly introduced.

It was postulated that full delineation of the properties discussed in Chapter 2 might not be necessary for adequate definition of the contamination and remediation problem. Increased definition of the more critical properties could be achieved through grouping insignificant or irrelevant parameters into a few lumped values. Sensitivity analysis or stochastic modelling might then be used to determine the sensitivity of the regime to changes in those neglected properties.

Local recharge/discharge and solute transport were considered the central components of Figure 2.1 with respect to pump-and-treat remediation. Introduction of extraction (and possibly injection) wells would alter the local recharge/discharge element of this flowchart. A change in the hydrogeology would result from this; as would a subsequent change in solute transport patterns toward extraction points. Changes in other connected components as a result of this action would also be important, and might act to impair the projected efficiency of the pump-and-treat system. At least a qualitative and semi-quantitative understanding of all the interrelationships in this regime would therefore be important when designing and operating a pump-and-treat remediation system.

5.1.2 Pump-and-Treat Case Studies

Table 3.13 summarized the effectiveness of the pump-and-treat remediation cases presented in Chapter 3. This review gave examples of successful remediation projects, promising initial results (subsequent data needed), and unsuccessful remediation projects.

The Firestone, Woodland Park, Southern Ontario A, and Southern Ontario B cases were all considered successful remediation efforts. They had either met the cleanup objectives set for the project and were slated for closure, or the response to remedial efforts had resulted in nearly 100% contaminant mass removal and attainment of goals appeared imminent.

Several case studies required follow-up studies to confirm the promising initial results. These included: South Brunswick Township, San Francisco Bay Area, New England, and Atlantic Coastal Airport. Caution was observed in projecting goal attainment in these cases due to the typical asymptotic responses to pump-and-treat remediation, as described by Hockman (1992). Conditions in a few of these sites promoted expectations for continued success, while other sites had problematic conditions that encouraged conservatism in projections.

The Central New Jersey and Strasbourg-Entzheim Airport sites were considered failures for the purposes of this review. These projects showed either negative or virtually zero response to extraction efforts. In the case of the Strasbourg-Entzheim Airport, remediation efforts were discontinued in favour of well resource management. The highly problematic conditions at the sites, with respect to the contaminant transport regime, was postulated as the cause of these failures.

The Verona well field site was designed as a containment project only. It appeared to have succeeded in attaining its objectives. However, there were significant gaps in the data that would be required for interpretation of the project's responses and degree of success in contamination abatement.

The summary of these cases provided in Table 3.13 did not reveal an obvious relationship of success or failure to the contaminant transport regime configuration. A dominating characteristic of each site was postulated to be a major contributor to project results, though the nature of this characteristic varied between sites. Caution was suggested in assessing patterns from this sample group. The population of published case studies tends to be skewed toward successes, and therefore a fully representative cross-section of remediation projects was not expected. Also, the amount of detail provided for sampling and testing protocols in these cases was cited as lacking for full interpretation of data worth.

5.1.3 Characterization and Design Using Scientific and Empirical Data

The methodology for ground water remediation projects was presented in flowchart form (Figure 4.1) at the beginning of Chapter 4. It was suggested as a general guideline to the process followed for the case studies of Chapter 3. The interrelationship between this methodology and the contaminant transport regime described in Chapter 2 was then discussed.

The terminology of inference, prediction and direct characterization was defined. Numerical modelling was then discussed in detail as an aid to these processes, with particular attention paid to inverse modelling techniques. A listing of commercial models for various contaminant transport processes and their capabilities was presented in Table

4.1. The issue of coupling capability between modelled processes was introduced as a method of more complete and realistic modelling of the regime.

Ground water flow and solute transport processes were coupled in a numerical modelling effort involving one of the more detailed case studies of Chapter 3. The three stages covered in that case allowed some degree of inverse modelling procedures to be applied. That is, plume development, interim action, and initial remediation project responses.

The modelling effort was proven to produce a working conceptualization for the case in question with a limited supply of site characterization data. However, the lack of initial and follow up data was revealed to limit the scope and value of the results.

5.2 CONCLUSIONS

5.2.1 State of Practice

The introduction of this report cited several authorities in ground water remediation who had stated reservations about the possibility of goal achievement in contamination remediation projects. Travis and Doty (1990), Giampaoli and Bugo (1991), Freeze and Cherry (1989), and Bredehoeft (1992) have all contributed commentaries that critically examined the ability of pump-and-treat remediation technology to clean up contaminated aquifers and soil volumes. The limited number of documented successes in the field, encapsulated by the review of Chapter 3, support these conclusions and has driven many to re-examine the direction the subsurface remediation industry has taken in the past few decades.

A more recent review of the state of practice in ground water remediation was presented by Cherry *et al.* (1993), focusing particularly on immiscible contaminants such as DNAPLs. They introduced the following levels of goals for remediation projects: full containment, full aquifer restoration, and partial aquifer restoration. The partial aquifer restoration option involved containment of the source volume and removal of the plume through pump-and-treat action or natural flushing. The critical component for success in full aquifer restoration, as stated in this paper, was source mass removal.

The problematic nature of source mass removal has become common knowledge in the remediation field. This is especially true of DNAPL in the subsurface, and less so but still significant with LNAPL in some regimes. The review in Chapter 3 examined cases of each of these types of contamination (6 DNAPL, 5 LNAPL cases). It was evident in all of these cases that the degree of removal of the immiscible source masses was largely responsible for the success or failure of the project. An excellent illustration of this effect for LNAPL contamination was the differences in response between a relatively easy source removal case (Southern Ontario Site B) and a problematic source removal case (Strasbourg-Entzheim); where Southern Ontario B was considered a success to the point of closure while Strasbourg-Entzheim remedial efforts were abandoned in favour of supply well management.

Many of the reviews of pump-and-treat effectiveness have suggested that enhanced pump-and-treat techniques might be the only chance of reversing the low success rates found in field cases (Cherry *et al.*, 1993; Hasbach, 1993; Hockman, 1992; Mercer and Cohen, 1990; MacKay and Cherry, 1989). Suggestions have ranged from pump-and-treat as a containment system for the source volume while degradation processes reduce contaminant mass, to direct use of the extraction/injection system to introduce solubility modifying solutions that dramatically increase the mass recovery rate (or immobilize the mass). To this end, both bioremediation and surfactant enhanced flushing technologies have been popular areas of research in recent years. However, with these innovative methods the risk of worsening the ground water contamination problem must always be carefully considered. Toxic degradation products from biodegradation or remobilized source masses from surfactant flushing are two examples of negative effects to attempted clean-up enhancement.

MacKay and Harman (1993) provided a brief review of chlorinated solvent contaminated sites' response to pump-and-treat remediation project that was similar in direction to the more extensive case study examination of Chapter 3. Their paper followed the goal level methodology of Cherry *et al.* (1993) mentioned above and emphasized the benefits of quick source containment action on the success of partial aquifer restoration attempts. The nature of the subsurface regimes that has been responsible for lack of pump-and-treat success was identified as a probable hindrance to enhanced technologies as well. Physical or hydraulic isolation of the source zone was concluded to be the only viable method of contamination control in this paper.

One general conclusion can be gained from this evaluation of pump-and-treat remediation technology and methodology, and might apply to many other methods of remediation as well. That is, the success or failure of aquifer remediation efforts is most consistently influenced by two aspects of the site: the degree of removal of contaminating source material from the subsurface; and the heterogeneity of soil properties in the area to be remediated. Without effective removal of the source of contamination, pump-and-treat efforts cannot realistically be expected to accomplish more than containment. Also, heterogeneous distribution of soil properties within the aquifer, particularly those properties relating to ground water flow or NAPL distribution, has a detrimental effect on the success (or efficiency) of contaminant mass withdrawal actions through pump-and-treat remediation.

5.2.2 Observational Method

The short list of case histories reviewed in Chapter 3 showed at least one common occurrence in pump-and-treat remediation projects. That is, design modification or complete redesign of remediation projects due to the encounter of unanticipated conditions during system installation or operation. In at least one case this encounter led to a complete re-evaluation of the projects direction and goals (Zilliox *et al.*, 1992). The inefficiency of this type of occurrence is self evident, leading to a search for improved methodology of design, construction and operation for remediation systems.

Morgenstern (1994) identified the possibility of applying the observational method to environmental geotechnics. This was largely due to a similarity in the uncertainties encountered in these projects and those found in more traditional geotechnical works. This paper also identified the need for robustness of design when applying the observational method. That is, the project design should have the ability to adapt to a significant range of possible conditions without excessive additional costs. The amount of involvement that various regulatory agencies commonly apply to environmental issues must be viewed as a possible hindrance to the application of the observational method in this field. A dependence on quantifying the risks of contamination problems for possible receptors might be seen as being in direct conflict with the precept of accepting uncertainty and managing it throughout the project.

Peck (1969) summarized the observational method in applied soil mechanics with the following steps:

a) "Exploration sufficient to establish at least the general nature, pattern and properties of the deposits, but not necessarily in detail.

b) Assessment of the most probable conditions and the most unfavourable conceivable deviations from these conditions. In this assessment geology often plays a major role.

c) Establishment of the design based on a working hypothesis of behaviour anticipated under the most probable conditions.

d) Selection of quantities to be observed as construction proceeds and calculation of their anticipated values on the basis of the working hypothesis.

e) Calculation of values of the same quantities under the most unfavourable conditions compatible with the available data concerning the subsurface conditions.

f) Selection in advance of a course of action or modification of design for every foreseeable significant deviation of the observational findings from those predicted on the basis of the working hypothesis.

g) Measurement of quantities to be observed and evaluation of actual conditions.

h) Modification of design to suit actual conditions."

He stated that these steps could be followed to varying degrees, dependent on the exact nature of the site conditions and work intended. The paper made clear differentiation between methods that apply only part of the above listing and the observational method. A common example of this would be those projects that monitor behaviour and adapt their design accordingly during construction or operation. The lack of a specific design contingency made prior to observations that anticipated the possibility of these deviations excludes this example from being considered an application of the observational method.

Peck also made clear the significance of unanticipated deviations from the working hypothesis and their negative impact on the project. The selection of significant observations was suggested to be equally important. Responses in the dynamic design under the observational method must be tied to measurable conditions in order to be implemented at the appropriate time. This selection of observation parameters and values must be done with consideration given to the reliability of measurements in both the local and global sense. That is, the measurement must be evaluated as to whether it is an accurate representation of the subsurface conditions at the point of measurement; and whether the value is representative of the processes acting in the site as a whole.

Any discussion of the benefits and drawbacks of the observational method must consider the possibilities of: increased cost due to the dynamic design configuration, possible extended time of installation as observed responses are awaited, and contractual difficulties due to lack of a fully defined project at the outset. Also, additional cost for the increased expertise necessary for the personnel monitoring data and making response decisions should be considered. However, the possible and even probable benefits of this method in terms of final cost, or even in achievement of goals that were otherwise unattainable, might be anticipated to outweigh these drawbacks.

An example of application of the observational method to a feasibility study was provided in Mark *et al.* (1989). However, in this case the most unfavourable combination of conditions was not incorporated as the worst case scenario. This would have resulted in a project design with excessive capacity for an unrealistic required response. Rather, a maximum credible deviation was used in predefining responses. This paper also discussed the difficulty in selecting observed values to be assigned response actions under the observational method.

The issue of maximum credible deviation introduced in the above paper brings another aspect into the discussion. That is, the growing use of stochastic theory in characterizing ground water flow and contaminant transport regimes might be incorporated into generating some system of defining the term 'credible' quantitatively by probability of occurrence.

It should be remembered at this point that the observational approach is only a framework. Tools such as engineering judgment, numerical modelling, and statistical techniques are needed for successful implementation of the methodology (Smyth and Quinn, 1991). The preceding chapters of this report have supplied extensive overviews of modelling and statistical techniques in the fields of geologic characterization, ground water flow, and contaminant transport. Engineering judgment, however, cannot be captured in text; it must be gained through experience of the realities constantly encountered in field application of engineering works.

Peterson *et al.* (1983) and Singh *et al.* (1983) showed that early on in the history of the Superfund era there was a recognized need for flexibility or 'robustness' of project design. Though both these cases were largely excavate-and-dispose projects, this early acknowledgment of uncertainty in remediation scope is very significant.

The same general recognition of the need for an adaptive design methodology was brought forward in a much more recent paper by Hoffman (1993). Again, the full application of observational method precepts was not suggested, but the methodology used favoured that direction heavily. The major difference between Hoffman's methodology and the observational method was the lack of predesigned contingency plans for possible deviations. The 'smart pump-and-treat' method simply recognized the probability of deviation from the initial plan and subsequently implemented the installation in a phased manner, responding to the changes as they were encountered. This encouraged robustness of design but did not necessarily guarantee that a quick, efficient response could be made to encountered deviations. Failure to anticipate unfavourable conditions was cited as one of the critical errors in application of the observational method (Peck, 1969).

An interesting feature of this project was the description of progressive modelling sophistication as characterization detail increased. Starting initially with a simple model of linear flow velocity the project moved to semi-analytical transport models, and finally a set of two-dimensional numerical models. Three dimensional numerical models for flow and transport were planned for the future, as well as unsaturated flow and transport models. Another innovative feature was the suggested use of dynamic flow field management to minimize stagnant zones and physical retardation in the extraction volume.

Billa *et al.* (1993) also presented a phased approach to pump-and-treat system installation. In this paper they advocated quick initial action that could be incorporated into the full scale design at a later date. The concern in this case was that initial investigation data on the hydrogeologic properties and hydrostratigraphy seemed unrealistic. Caution was exercised by implementing a conservative treatment system with the pumping wells until the site responses could confirm characterization for the full scale operation. The results of this staged approach were considerable cost savings in what would have been an overdesigned treatment system.

One problem with lack of predesigned alternatives, as utilized in the several cases discussed above, is the lack of data for cost estimating in the initial stages. Cost-benefit evaluations of the various techniques and configurations is usually important in selecting the most economically efficient design that has the ability to attain project goals. From the point of view of the site owner, this advance knowledge of at least a range of costs for the project might be critical in alternative selection and budgeting procedures. Although the observational method cannot give one single cost estimate at the outset, it does allow for cost ranges and possibly even estimates of the probability for these ranges. Alternatively, single configuration designs may yield a single cost estimate, but this initial cost might be erroneous with additional components added as difficulties are encountered or overdesign resulting from inaccurate initial characterization.

5.2.3 Recommendations

The view that pump-and-treat in itself is incapable of remediating many sites has been suggested by many authorities in the field. It has also been suggested that enhanced pump-and-treat techniques, other innovative remediation technologies, isolation with ground water management, and more realistic goals might offer a better chance of improving the experiences in remediation than continued application of traditional technology. Freeze and Cherry (1989) postulated a retreat from the posture of universal cleanup and concentration of resources on sites where the resource is most critical and success attainable. Full extraction of contaminants from the ground over a few years of system operation when they have been migrating into the subsurface for decades is an unrealistic expectation (Hoffman, 1993).

The observational method and similar methodologies have been applied to a few cases over the history of site remediation (the past few decades). However, these few applications have yet to prove themselves to be more consistently successful and/or economical than traditional pump-and-treat techniques. Though the method seems tailor-made for environmental remediation, previously discussed shortcomings must be scrutinized in order to fully discern its usefulness. The observational method needs to be assessed more fully as projects where it has been applied move towards closure. Recommendations associated with the findings of Chapters 2, 3, and 4 that might act as enhancements to its successful implementation are presented in the following paragraphs.

Complexity of the contaminant transport regime was presented in Chapter 2 along with a discussion of various techniques for property and process determination. Some of these techniques involved analytical or numerical methods applied to modelling the various processes. One shortcoming of this characterization scenario was that few of the numerous interacting processes could be modelled in a coupled fashion. Also, if coupling of many of these processes was possible, the data and computational resources demands would make its use impractical. Advances in coupled modelling techniques might lead to much more accurate modelling of the regime when used with the iterative characterization development of the observational method for data supply.

The eleven case studies reviewed in Chapter 3 failed to show an identifiable association between geology of a site and pump-and-treat remediation response. Some tentative patterns were noted, however, and further examination of this phenomena might be warranted. A more extensive review of all available case studies, not only in literature but in project reports, might clarify the nature or existence of this speculated pattern. A relationship such as this, even of the most general type, could greatly aid in the application of pump-and-treat system design and especially the observational method.

Chapter 4 discussed a linking between the scientific knowledge of the contaminant transport field and the empirical database of pump-and-treat project behaviour. Application of this link has been experienced in many cases to some degree. Hoffman (1993) provided a good example of the increase in modelling detail based on the increased data supplied by a continuing project. A formalized methodology of this iterative relationship between site characterization and project responses would be an integral part of any application of the observational method to environmental remediation.

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APPENDIX A: Hydrogeologic and Geotechnical Soil Properties

Table A.1: Ranges of Soil Hydraulic Conductivity

Soil Type	**Hydraulic Conductivity**		**Reference**
	Range		
	low	high	
	(m/s)	(m/s)	
Karst Limestone	1.00E-06	1.00E-02	Freeze & Cherry (1979)
Permeable basalt	1.00E-07	1.00E-02	
Fractured i&m rock	1.00E-08	1.00E-04	
Limestone & dolomite	1.00E-09	1.00E-06	
Sandstone	1.00E-10	1.00E-06	
Unfractured i&m rock	1.00E-14	1.00E-10	
Shale	1.00E-13	1.00E-09	
Marine clay (unweathered)	1.00E-12	1.00E-09	
Glacial till	1.00E-12	1.00E-06	
Silt, loess	1.00E-09	1.00E-05	
Silty sand	1.00E-07	1.00E-03	
Clean sand	1.00E-05	1.00E-02	
Gravel	1.00E-03	1.00E+00	
River deposits			Lambe & Whitman (1969)
Rhone	less than	4.00E-03	
small streams, eastern Alps	2.00E-04	1.60E-03	
Missouri	2.00E-04	2.00E-03	
Mississippi	2.00E-04	1.20E-03	
Glacial deposits			
outwash plains	5.00E-04	2.00E-02	
esker	1.00E-04	1.30E-03	
delta	1.00E-06	1.50E-04	
till	less than	1.00E-06	
Wind deposits			
dune sand	1.00E-03	3.00E-03	
loess	approx.	1.00E-05	
loess loam	approx.	1.00E-06	
Lacusterine and marine offshore deposits			
v. fine, uniform sand	1.00E-06	6.40E-05	

Table A.2: Aquifer Properties - Field Data & Case Studies

Soil Type, Aquifer	Hydraulic Conductivity		Transmissivity		Test Type	Hydraulic Gradient	Groundwater Velocity	Storativity	Reference
	Range		Range					[Specific Yield]	
	low	high	low	high					
	(m/s)	(m/s)	(m^2/s)	(m^2/s)		(m/s)	(m/s)		
old bridge sand	7.55E-04	1.65E-03			pumping test				Roux and Althoff (1980)
sand-gravel channel deposits: unconfined	-	3.47E-04			pumping tests				Smedes et al. (1993)
sand-gravel channel deposits: confined	6.94E-04	4.17E-03			pumping tests				
fine to coarse sands w/ silt, gravel and clay	8.11E-07	9.88E-06			slug test				Herkert and Aslam (1990)
	3.35E-06	1.66E-05			residual drawdown test				
	1.33E-04	2.20E-04			24 hour pump test				
uniform fine sands	-	1.76E-05			-				Duke (1987)
clayey & silty sands	less than	3.53E-06			-				
silty sands, fine sands	-	3.53E-06			-				
fine sands	-	1.06E-05			-				
fine sand/peat/silty sand/weathered limestone	-	7.06E-05			-				
silty sands and gravels/ clay	2.70E-07	5.10E-06			rising head permeability				Donovan and Murray (1985)
high yield glacial aquifer	-	2.83E-03			8 hour pumping test				Penzo and Boyer (1986)
Rhine alluvial deposits	-	2.00E-03			single well test				Zilliox et al. (1992)
sand and gravel glacial outwash	-	1.27E-03			natural gradient tracer test	0.0015	1.41E-06		Garabedian and LeBlanc (1990)
poorly to well-sorted sandy gravel/gravelly sand; shallow alluvial terrace deposit, Kh	2.20E-05	2.00E-04			large scale aquifer tests	0.003		[0.04 - 0.10]	Boggs and Rehfeldt (1990)
ibid, Kv	1.30E-06	2.80E-05			large scale aquifer tests				

Table A.2: Aquifer Properties - Field Data & Case Studies

Soil Type, Aquifer	Hydraulic Conductivity		Transmissivity		Test Type	Hydraulic Gradient	Groundwater Velocity	Storativity	Reference
	Range		Range					[Specific Yield]	
	low	high	low	high					
	(m/s)	(m/s)	(m^2/s)	(m^2/s)		(m/s)	(m/s)		
	-	2.50E-05			tracer injection test				
allcvial aquifer	-	3.20E-04			b.h. flowmeter				Young (1990)
Del Park catchement, Western Australia	less than	8.78E-06							Peck et al. (1988)
Chicot aquifer, Texas			less than	[illegible]				0.0005 - 0.1	
Cenomanian aquifer, Beirut, Lebanon			1.00E-02	[illegible]				0.01 - 0.03	
San Jose del Cabo Valley, California			1.39E-03	2.50E-01				[0.13 - 0.22]	
Candover catchment, Hampshire, UK			9.26E-03	9.26E-02					
Severouralsk karst basin, USSR	1.16E-05	1.16E-03							
Snake River Plain aquifer, Idaho			4.00E-03	*1.40E+01*				*0.001 - 2*	
Houston/Trinity aquifer, Texas			1.39E-04	5.79E-03					
Sabi Valley, Zimbabwe			*1.70E+02*	*4.30E+03*				0.0001 - 0.14	
Jordan aquifer, Iowa			5.32E-04	1.02E-02					
Animas Valley, New Mexico			3.13E-03	3.54E-02				0.07 - 0.14	
Hyderadad, Andhra Pradesh State, India			1.04E-04	2.65E-03				0.01 - 0.04	
Eastern perimeter, Bighorn artesian basin, Wyoming			2.89E-06	6.37E-04					
Oak Ridge Reservation, Tennessee			6.02E-06	5.09E-05				0.0029 - 0.014	
Ajali aquifer, Nigeria	2.22E-06	6.06E-04							
Fairfax-New Baltimore aquifer, Ohio	3.33E-03	2.25E-02						0.002 - 0.2	

Table A.2: Aquifer Properties - Field Data & Case Studies

Soil Type, Aquifer	Hydraulic Conductivity		Transmissivity		Test Type	Hydraulic Gradient	Groundwater Velocity	Storativity	Reference
	Range		Range					[Specific Yield]	
	low	high	low	high					
	(m/s)	(m/s)	(m^2/s)	(m^2/s)		(m/s)	(m/s)		
Upper Floridan aquifer, Volusia County, Florida			4.28E-03	5.38E-02					
Deccan Trap Basalts, Madhya Pradesh State, India			1.50E-04	3.70E-03					
			Theis	M & C					Booth (1988)
Sandstone aquifer - Saltsburg			2.73E-04	1.03E-03	Brief pumping well tests				
Sandstone aquifer - Morgantown			9.14E-05	2.46E-05					
Sandstone aquifer - Saltsburg			1.87E-05	1.60E-04					
Sandstone aquifer - Morgantown			5.30E-05	5.48E-05					
Sandstone aquifer - Saltsburg			2.00E-05	5.59E-05					
Sandstone aquifer - Saltsburg			2.04E-05	n/a					
Shales			3.45E-06	2.73E-06					
	Slug	CPT							Smythe et al. (1989)
clay aquitard	1.52E-08	1.72E-08							
	5.33E-08	2.84E-08							
	Slug	Lab							
quartz sand	2.40E-04	2.20E-04							
	2.40E-04	2.60E-04							
poorly sorted sand and gravel	2.50E-05	1.00E-04			slug tests				Jackson et al. (1985)
med. to coarse sand and gravel w/ cobbles	1.30E-04	1.20E-03			long term pumping tests				
	avg.	8.10E-05			slug tests				
dense, fine grained silty sand and sandy silt w/some thin clay beds	-	5.00E-06			slug test				
stratified sand and gravel beds	avg.	3.10E-04			pump tests				

Table A.2: Aquifer Properties - Field Data & Case Studies

Soil Type, Aquifer	Hydraulic Conductivity		Transmissivity		Test Type	Hydraulic Gradient	Groundwater Velocity	Storativity	Reference
	Range		Range					[Specific Yield]	
	low	high	low	high					
	(m/s)	(m/s)	(m^2/s)	(m^2/s)		(m/s)	(m/s)		
	1.00E-05	1.00E-04			slug tests				
fine to coarse sand and landfill material	5.00E-07	6.10E-06			pump tests / slug test respectively				
outwash aquifer - poorly sorted gravels, sands, and silts	avg.	1.10E-04			slug tests				
stratified clayey silt and silt	avg.	8.00E-06			slug tests				
regressional sands and gravels	avg.	4.00E-05			slug tests (K), point dilution (v)		2.89E-07		
outwash aquifer - poorly sorted gravels, sands, and silts	6.00E-05	1.70E-03	1.20E-03	3.40E-02	pump tests (K and T), point dilution tests (velocity)		6.37E-07	2.50E-03	
	mean	4.90E-04	mean	9.80E-03					
outwash aquifer - poorly sorted gravels, sands, and silts					bomb tritium tracer tests		5.79E-07		Jackson et al. (1985)
silty clay	1.00E-10	1.00E-11			consolidation tests				Schmidtke et al. (1992)
silty sand		1.00E-05			slug and pumping tests				
silty clay		1.00E-10			consolidation tests				
glacial till (clayey silt) aquifer	2.30E-07	8.50E-06			slug tests	0.02	2.85E-07		Meiri et al. (1990)
glacial till (silty clay) aquitard, (Kv)	3.00E-10								
weathered shale aquifer		3.00E-06		1.50E-05	drawdown/leaky	0.01		3.00E-04	
weathered shale aquifer				5.79E-06	drawdown data/Jacob method				
weathered shale aquifer				3.47E-06	recovery data				
Borden site	6.00E-06	2.00E-04			lab core testing (temp. corrected)	0.0043	1.04E-06		Sudicky (1986)
Columbus AFB						0.0015	4.86E-06		Hess et al. (1992)

Table A.2: Aquifer Properties - Field Data & Case Studies

Soil Type, Aquifer	Hydraulic Conductivity		Transmissivity		Test Type	Hydraulic Gradient	Groundwater Velocity	Storativity	Reference
	Range		Range					[Specific Yield]	
	low	high	low	high					
	(m/s)	(m/s)	(m^2/s)	(m^2/s)		(m/s)	(m/s)		
SEE Gelhar (1992)								**Dispersivity (Al/At/Av)**	Gelhar et al. (1992)
Columbus, very hetero. sand and gravel (Adams and Gelhar 1991)	1.00E-05	1.00E-03			ambient flow field		3.5e-7 to 5.8e-6	7.5	
Hanford, glaciofluvial sands and gravels (Ahlstrom et al. 1977)	5.70E-04	3.00E-02			ambient		3.47E-06	30.5 / 18.3	
Hanford (Bierschenk 1959; Cole 1972)				1.70E-01	ambient		3.01E-04	6 to 460	
Brunswick, limestone (Bredehoeft and Pinder 1973)			6.50E-07	8.60E-07	radial converging			170 / 52	
Amargosa, fractured dolomite and limestone (Claasen and Cordes 1975)			5.00E-02	1.10E-01	2 well recirculating		1.6e-6 to 3.9e-5	15	
Nevada, alluvium derived from tuff (Daniels 1981/82)		1.70E-05			radial converging		4.63E-07	10 to 30	

Table A.3: Stochastic Properties of Aquifers

Property	Soil Type	Mean	Variance	Coeff. of	Correlation horiz./vert.	Reference
porosity	clay loam	0.45	2.45E-03	11%		Jury (1986)
	sand	0.37	1.66E-03	11%		
	clay loam	0.53	1.38E-03	7%		
	loamy sand	0.42	1.76E-03	10%		
porosity	outwash aquifer - poorly sorted gravels, sands, and silts	0.30 to 0.35				Jackson et al. (1985)
	regressional sands and gravels	0.35 to 0.40				
	stratified clayey silt and silt	0.40 to 0.45				
Porosity	sandy gravel / gravelly sand (poorly to well sorted)	0.31	6.40E-03	26%		Boggs and Rehfeldt (1990)
bulk density (g/cm^3)	clay loam	1.36	9.06E-03	7%		Jury (1986)
	sandy loam	1.3	8.28E-03	7%		
	sandy loam	1.2	9.73E-02	26%		
	sandy loam	1.47	1.75E-02	9%		
	silty clay	1.26	5.72E-03	6%		
	silt loam	1.47	7.78E-03	6%		
	sand	1.65	2.45E-03	3%		
	sand	1.59	9.10E-03	6%		
	clay loam	1.2	3.24E-02	15%		
Bulk density (g/cm^3)	sandy gravel / gravelly sand (poorly to well sorted)	1.77	3.24E-02	10%		Boggs and Rehfeldt (1990)
Particle density (g/cm^3)	sandy gravel / gravelly sand (poorly to well sorted)	2.57	4.90E-03	3%		Boggs and Rehfeldt (1990)
% sand / % clay	clay	24 / 45		15 / 33%		Jury (1986)
	silty clay loam	7 / 32		32 / 16%		
	sandy loam	[illegible]		37 / 53%		

Table A.3: Stochastic Properties of Aquifers

Property	Soil Type	Mean	Variance	Co of	Correlation	Reference
					horiz./vert.	
	loamy sand	83 / 9		3 / 34%		
	sandy clay loam	65 / 28		65 / 28%		
% sand	sandy clay laom				36	Jury (1986)
	sand				30	Jury (1986)
% sand					30 / 40	Warrick and Yeh (1989)
% sand					5	
% sand					35	
organic carbon					< 8	Warrick and Yeh (1989)
organic carbon fraction	regressional sands and gravels	0.15%	0.01%	67%		Jackson et al. (1985)
	outwash aquifer - poorly sorted gravels, sands, and silts	0.46%	0.08%	60%		Jackson et al. (1985)
	stratified clayey silt and silt	0.41%	0.02%	30%		
sorptivity					37	Warrick and Yeh (1989)
Kd (cm^3/g)	loamy sand	2.01	3.88E-01	31%		Jury (1986)
Rf	Palo Alto Baylands (with CHCl3)	3.2	3.60E-01	19%		Jackson et al. (1985)
	Palo Alto Baylands (with CHBr3)	6				
	Palo Alto Baylands (with Cl3CCH3))	12				
	Palo Alto Baylands (with C6H5Cl)	33				
	Borden (with CHBr3)	1.7	4.00E-02	12%		
	Borden (with CCl4)	1.6	4.00E-02	13%		
Rf	outwash aquifer - poorly sorted gravels, sands, and silts (with 1,4-	1.6				Jackson et al. (1985)
	outwash aquifer (with Tetrahydrofuran)	2.2				

Table A.3: Stochastic Properties of Aquifers

Property	Soil Type	Mean	Variance	Coeff. of	Correlation	Reference
					horiz./vert.	
	outwash aquifer (with Diethyl ether)	3.3				
	outwash aquifer (with 1,2-Dichloroethane)	7.6				
	outwash aquifer (with Benzene)	8.8				
	outwash aquifer (with Carbon tetrachloride)	23				
water content					< 16	Warrick and Yeh (1989)
water content					76	
water content (@1bar)	clay loam	37.0%	2.19E-04	4%		Jury (1986)
	clay loam	37.0%	4.24E-03	18%		
	sandy loam	27.0%	2.92E-03	20%		
	clay loam	45.0%	4.56E-03	15%		
water content (@15 bar)	clay loam	16.6%	5.71E-04	14%		
	sandy loam	4.1%	3.40E-04	45%		
	silty clay	19.3%	7.30E-04	14%		
	silt loam	7.4%	1.98E-04	19%		
	sandy loam	9.5%	9.83E-04	33%		
Sr - water	various	17.5%	3.36E-03	33%		Mercer and Cohen (1990)
Sr - DNAPLs	various	19.7%	3.10E-03	28%		
infiltration rate					0.13	Warrick and Yeh (1989)
infiltration rate					50	Warrick and Yeh (1989)
infiltration rate (m/s)	clay loam	1.69E-06	2.52E-12	94%		Jury (1986)
	loam	1.94E-06	6.05E-13	40%		
	silty clay loam	7.64E-07	2.94E-13	71%		
	silty clay	9.84E-07	3.04E-13	56%		

Table A.3: Stochastic Properties of Aquifers

Property	Soil Type	Mean	Variance	Coeff. of	Correlation	Reference
					horiz./vert.	
	silty clay loam	9.84E-07	5.12E-14	23%		
	7 series	5.44E-06	1.85E-11	79%		
	7 series	3.04E-05	8.72E-10	97%		
infiltration rate	clay loam				0.13	Jury (1986)
	weathered shale				<2.0	
	sandy loam				35	
Ksat (m/s)	clay loam	2.38E-06	8.19E-12	120%		Jury (1986)
	sandy loam	1.94E-05	1.36E-09	190%		
	sand	3.66E-05	6.37E-10	69%		
	loamy sand	9.72E-06	4.50E-11	69%		
	silty clay loam	4.17E-07	4.00E-14	48%		
	coarse	2.15E-06	4.92E-12	103%		
	fine	1.27E-06	2.26E-12	118%		
	silty clay	7.99E-07	5.40E-13	92%		
	very coarse	3.25E-06	1.08E-10	320%		
	coarse	6.44E-06	5.77E-11	118%		
	loamy sand	8.24E-06	7.49E-11	105%		
	loamy sand	1.14E-05	8.53E-11	81%		
	sandy loam	2.79E-06	2.47E-11	178%		
	silt loam	2.35E-05	1.38E-10	50%		
Ksat	sand				25m	Jury (1985)
	-				1.6	
Ksat (log-normal)	conglomerate	-4.59	0.94	21%		Freeze (1975)
	sandstone	-5.38	0.61	15%		
	marly limestone	-5.38	0.46	13%		
	vuggy limestone	-5.60	0.53	13%		

Table A.3: Stochastic Properties of Aquifers

Property	Soil Type	Mean	Variance	Coeff. of	Correlation	Reference
					horiz./vert.	
	sandstone	-3.36	0.20	13%		Freeze (1975)
	sandstone	-3.82	0.40	17%		
	sandstone	-4.60	0.40	14%		
	sandstone	-5.46	1.00	18%		
	sandstone	-5.52	0.65	15%		
	sandstone	-4.60	0.32	12%		
	sand and gravel		0.44			
	sand and gravel		0.72			
	sand and gravel		0.54			
	clay loam	-5.41	0.45	12%		
	clay loam	-4.00	0.85	23%		
	clay loam	-4.60	0.93	21%		
	silty clay	-4.96	0.78	18%		
	silty clay	-4.46	1.56	28%		
	silty clay	-3.00	0.93	32%		
	loamy sand	-2.68	0.86	35%		
Ksat					16	Hoeksema and Kitanidis (1985)
Ksat					34	Warrick and Yeh (1989)
Ksat (log(m/d))	interchannel sediments	-2.73	1.69	48%		Fogg (1986)
	channel-fill sands	0.58	0.13	62%		
Ksat (ln-normal) -	sand and gravel - glacial outwash	-6.91	0.26	7%	5.1 / 0.26	Hess (1989)
Ksat (ln-normal) -	sand and gravel - glacial outwash	-7.96	0.14	5%		Hess (1989); Hess et al. (1992)
K variability (lnK)	sand and gravel - glacial outwash		0.24		2.6 / 0.19	Hess (1990) (see Garabedian 1990)
K variability	Aefligen (Hufschmied 1986)	0.36	2.15	407%	15-20 / .05-	Hess et al. (1992)

Table A.3: Stochastic Properties of Aquifers

Property	Soil Type	Mean	Variance	Coeff. of	Correlation	Reference
					horiz./vert.	
Ksat (ln-normal)	sandy gravel / gravelly sand (poorly to well sorted)	-11.5	2.8	15%		Boggs and Rehfeldt (1990)
Ln(K)	sandy gravel / gravelly sand (poorly to well sorted)		4.5		12.0 / 1.5	Boggs and Rehfeldt (1990)
	ibid (detrended data)		2.8		5.3 / 0.7	
	ibid (grain size K)		2.9		15 / 0.8	
	ibid (slug test)		1.8			
	ibid (permeameter)		5.5			
	glaciofluvial		0.8			
	fluvial sands		0.9			
	alluvial sand and gravel		1.5 - 3.7			
ln(K)	Borden Site (Sudicky 1986)		0.38		2.8 / 0.12	Gelhar (1993)
	Borden Site (Woodbury and Sudicky 1991)		0.24		5.1 / 0.21	
	Borden Site (Woodbury and Sudicky 1991)		0.37		8.3 / 0.34	
	Borden Site (Robin et al. 1991))		0.37		1-7.5 / 0.1-0.3	
ln(K)	Borden site	-9.54301981	0.29	6%	2.8 / 0.12	Sudicky (1986)
K (m/s)		(geometric)	var[ln(K)]			Young (1990)
	alluvial aquifer (pump test, low rate)	3.20E-04	4.7	27%		
	alluvial aquifer (pump test, low rate, S=.1)	4.00E-05	4.2	20%		
	alluvial aquifer (pump test, low rate, S=.001)	6.00E-05	4.1	21%		
	alluvial aquifer (pump test, low rate, S=.00001)	8.00E-05	4.1	21%		

Table A.3: Stochastic Properties of Aquifers

Property	Soil Type	Mean	Variance	Coeff. of	Correlation	Reference
					horiz./vert.	
ln(Ksat)	sandstone aquifer		2.3 to 4.8		- / 0.3-1.0	Gelhar (1993)
	fluvial sand		0.81		> 3 / 0.1	
	fluvial soil		1.00		7.6	
	eolian sandstone outcrop		0.16		8 / 3	
	glacial outwash sand		0.25		5 / 0.26	
	sand and gravel aquifer		3.61		20 / 0.5	
	prairie soil		0.36		8	
	weathered shale subsoil		0.64		< 2	
	fluvial sand and gravel aquifer		4.41		13 / 1.5	
	Hama red mediterraenean soil		0.16 to 1.2		14 to 39	
	gravelly loamy sand soil		0.49		500	
	alluvial silty-clay loam soil		0.36		0.1	
	glacial outwash sand and gravel outcrop		0.64		5 / 0.4	
	glacial-lacusterine sand aquifer		0.36		3 / 0.12	
	alluvial soil (Yolo)		0.81		15	
ln(Ksat)	various	-8.78	5.79	27%		Gelhar et al. (1992)
intrinsic permeability	silty sand layer (from DNAPL data)	8.60E-13	2.57E-25	59%		Schmidtke et al. (1992)
	silty sand layer (from APL data)	1.21E-12	1.82E-24	112%		
Tracer tests						
Dispersivity (L/Th/Tv)	sand and gravel - glacial outwash	0.96/.018/				Garabedian and LeBlanc (1990)
Dispersivity (L/T) (m)	Borden site (Freyberg 1986)	0.36 / 0.039				Hess et al. (1992)
Dispersivity (L) (m)	Aefligen (Berdat et al. 1986)	7.8 to 22				
Dispersivity (L) (m)	Columbus AFB (Young and Boggs 1990)	10 to 42				
log(T)					9600	Warrick and Yeh (1989)

Table A.3: Stochastic Properties of Aquifers

Property	Soil Type	Mean	Variance	Coeff. of	Correlation	Reference
					horiz./vert.	
Transmissivity	consolidated				17 km	Hoeksema and Kitanidis (1985)
	unconsolidated				28	
Transmissivity (m^2/s)			var(T)			Young (1990)
	alluvial aquifer (aq. test 1)	3.70E-03	4.90E-07	19%		
	alluvial aquifer (aq. test 2)	3.20E-03	8.10E-07	28%		
	alluvial aquifer (aq. test 3)	3.30E-03	1.69E-06	39%		
	alluvial aquifer (pump test, high rate)	3.80E-03	4.00E-06	53%		
	alluvial aquifer (pump test, low rate)	7.50E-03	9.41E-05	129%		
	alluvial aquifer (slug test)	4.00E-04	4.00E-08	50%		
	alluvial aquifer (pump test, low rate, S=.1)	8.00E-04	4.90E-07	88%		
	alluvial aquifer (pump tes, low rate, S=.001)	1.10E-03	1.00E-06	91%		
	alluvial aquifer (pump test, low rate, S=.00001)	1.50E-03	1.69E-06	87%		
Transmissivity (m^2/s)	silty sand lay	4.25E-05	1.80E-10	32%		Schmidtke et al. (1992)
ln(T)	alluvial-basin aquifer		1.49		4000	Gelhar (1993)
	alluvial-basin aquifer		1.00		800	
	limestone aquifer		5.29		6300	
	sandstone aquifer		1.96		17000	
	alluvial aquifer		0.36		150	
	alluvial aquifer		0.16		1800	
	limestone aquifer		5.29		3500	
	chalk		2.89		7500	
	alluvial aquifer		0.64		820	

Table A.3: Stochastic Properties of Aquifers

Property	Soil Type	Mean	Variance	Coeff. of	Correlation	Reference
					horiz./vert.	
	sandstone aquifer		0.36		4.50E+04	
ln(T)	various	-5.56	18.68	78%		Gelhar et al. (1992)
velocities (log-var.)	loamy sand		0.48			Jury (1986)
	clay		0.32			
	loam		1.56			
velocities	various	1.77E-04	9.97E-08	178%		Gelhar et al. (1992)
ln(velocities)		-10.34	6.25	24%		
S coeff.					24 (min)	Hoeksema and Kitanidis (1985)
Storativity	silty sand layer	0.011	1.21E-04	100%		Schmidtke et al. (1992)
P sorbed					32000	Warrick and Yeh (1989)
P sorbed					58000	Warrick and Yeh (1989)
chemical concentrations	loamy sand (Cl) - 0.65m depth			127%		Jury (1986)
	loamy sand (Cl) - 1.15m			79%		
	loamy sand (Cl)			76%		
	loamy sand (Br) - 0.3m			118%		
	loamy sand (Br) - 0.6m			90%		
	loamy sand (Br) - 0.9m			89%		
	sandy loam (Br)			100%		
	loamy sand (Cl) - 0.12m			67%		
	loamy sand (Cl) - 0.32m			61%		
	Mankos shale (EC) - 0 to .025m			128%		
	Mankos shale (EC) - .075 to .15m			263%		
	clay loam-loam (EC) - 0 to 1m			91%		
	clay loam-loam (EC) - 0 to 1m			75%		
	clay loam-loam (EC) - 0 to 1m			225%		
	sandy loam-loam (Cl) - 1.8 to 6m			22%		

Table A.3: Stochastic Properties of Aquifers

Property	Soil Type	Mean	Variance	Coeff. of	Correlation	Reference
					horiz./vert.	
	sandy loam-loam (Cl) - 1.8 to 6m			30%		
	loam - sandy loam (Cl) - 1 to 3.6m			101%		
	loam - sandy loam (Cl) - 1 to 3.6m			21%		
	various (Cl) - surface to bedrock			12-70%		
	sandy loam (Cl) - 1.5 to 6.3m			19%		
	sandy loam (Cl) - 1.5 to 6.3m			19%		
	clay loam (Cl) - 1.5 to 6.3m			66%		
	loam (Cl) - 0.6 to 1.2m			60%		
	fine sandy loam (Cl) - 0.9 to 1.5m			102%		
	fine sandy loam (Cl) - 0.6 to 1.2m			87%		
EC	clay loam				800	Jury (1986)
	clay loam				1.2	
	clay loam				20	
	clay loam				20	
log(EC)					790 m	Warrick and Yeh (1989)
concentration (Na)					61	

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APPENDIX B: Stage 2 Modelling Input Files and Data

Table B.1: Input ASCII Files for MODFLOW Model - Stage 2

SBT2BAS.DAT

```
CASE STUDY, STEADY STATE, REDUCED PUMPING ANALYSIS
SOUTH BRUNSWICK TOWNSHIP
         1        46        39         1         4
 11 12 00  0  0  0  0 18 19  0  0 22 00 00 00 00 00 00 00 00 00 23 00 00
         1         0
         1         1(39I3)                       1
 -1  1 -1 -1 -1 -1 -1 -1 -1 -1 -1 -1 -1 -1 -1 -1 -1 -1 -1 -1 -1 -1 -1 -1 -1 -1 -1 -1
 -1 -1 -1  0  0  0  0  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1 -1 -1  1 -1 -1 -1  1  1  1  1
  1  1  1 -1  0  0  0  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1 -1  1  1  1  1  1  1  1
  1  1  1 -1  0  0  0  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1 -1  0  0  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1 -1  0  0  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1 -1  0  0  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1 -1  0  0  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1 -1  0  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1 -1  0  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1 -1  0  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1 -1  0  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1 -1  0  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1  1 -1  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1  1 -1  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1  1 -1  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
  1  1 -1  1  1  1 -1  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
  1  1 -1  1  1  1 -1  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
  1  1 -1 -1  1  1  1 -1  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
  1  1  1 -1  1  1  1 -1  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
  1  1  1 -1  1  1  1 -1  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
  1  1  1 -1  1  1  1 -1  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
  1  1  1 -1 -1  1  1  1 -1  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
  1  1  1 -1 -1  1  1  1 -1  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
  1  1  1 -1 -1  1  1  1  1 -1  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
  1  1  1 -1 -1  1  1  1  1 -1  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
  1  1  1 -1 -1  1  1  1  1 -1  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
  1  1  1 -1 -1  1  1  1  1  1 -1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
  1  1  1 -1 -1  1  1  1  1  1 -1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
  1  1  1 -1 -1  1  1  1  1  1 -1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
  1  1  1 -1 -1  1  1  1  1  1 -1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1  1  1  1  1  1 -1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1  1  1  1  1  1 -1
```

```
  1  1 -1 -1 -1 -1 -1 -1 -1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1 -1
  1  1 -1 -1 -1 -1 -1 -1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1 -1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1 -1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1 -1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1 -1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1 -1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1 -1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1 -1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1 -1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1 -1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1 -1 -1 -1 -1 -1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
-1 -1 -1 -1 -1 -1  0  0  0  0  0
  1  1  1  1  1 -1 -1 -1 -1 -1 -1 -1 -1 -1 -1 -1 -1 -1 -1 -1 -1  1 -1  1  1  0  0  0
0  0  0  0  0  0  0  0  0  0  0
 -1 -1 -1 -1 -1  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0     0  0  0  0
0  0  0  0  0  0  0  0  0  0  0
      -99.
        99        0(13F8.1)                       -1
     18250         1         1.
```

SBT2BCF.DAT

```
         1        31
         1
         0        1.
        11        0.(14F6.0)                      -1
  200.  200.  200.  200.  200.  100.  100.  100.  100.   50.   50.   50.   50.   50.
   50.   50.   50.   50.   50.   50.   50.   50.   50.   50.   50.  100.  100.  100.
  100.  200.  200.  200.  200.  200.  200.  200.  200.  200.  200.
        11        0.(16F6.0)                      -1
  100.  100.  100.  100.   50.   50.   50.   50.   50.   50.   50.   50.   50.   50.
50.   50.
   50.   50.   50.   50.  100.  100.  100.  100.  100.  100.  100.  100.  200.  200.
200.  200.
  200.  200.  200.  200.  200.  200.  200.  200.  200.  200.  200.  200.        200.
       101         0                              -1
2
  1  5  1 39   150.
  6 46  1 39   350.
         0        60.
```

SBT2WEL.DAT

```
         1        31
         1
         1        10        18         0
```

SBT2RCH.DAT

```
         1        31
         1
         0     .0001
```

SBT2SIP.DAT

```
      100          5
       1.       1.E-4           1           1
```

SBT2OPC.DAT

```
    0       0      30       0
    0      30       0      31
    0       0      30       0
    0      30       0      31
    0       0      30       0
    0      30       0      31
    0       0      30       0
    0      30       0      31
    0       0      30       0
    0      30       0      31
    0       0      30       0
    0      30       0      31
    0       0      30       0
    0      30       0      31
    0       0      30       0
    0      30       0      31
    0       0      30       0
    0      30       0      31
    0       0      30       0
    0      30       0      31
    0       0      30       0
```

Table B.2: MODFLOW Model - Stage 2 Input Hydraulic Heads

	Column 1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20
Row 1	***91.00***	***90.00***	***89.00***	***88.00***	***87.00***	***87.50***	***88.00***	***88.50***	***89.00***	***89.50***	***90.00***	***91.00***	***92.00***	***93.00***	***94.00***	***95.00***	***95.50***	***96.00***	***96.00***	***96.00***
2	89.74	89.40	88.86	88.30	87.90	88.00	88.30	88.69	89.12	89.60	90.15	90.89	91.68	92.49	93.27	93.99	94.49	94.87	95.03	95.11
3	89.14	89.01	88.74	88.45	88.27	88.32	88.52	88.82	89.19	89.63	90.13	90.72	91.35	92.00	92.62	93.18	93.62	93.94	94.14	94.26
4	88.81	88.76	88.63	88.50	88.43	88.48	88.64	88.89	89.21	89.58	90.01	90.50	91.01	91.54	92.04	92.48	92.86	93.14	93.34	93.47
5	88.60	88.58	88.52	88.47	88.47	88.53	88.68	88.89	89.16	89.48	89.85	90.25	90.67	91.10	91.50	91.87	92.18	92.43	92.61	92.73
6	88.43	88.43	88.41	88.41	88.43	88.51	88.64	88.83	89.06	89.33	89.64	89.98	90.33	90.67	91.01	91.31	91.57	91.78	91.94	92.06
7	88.28	88.28	88.29	88.31	88.35	88.44	88.56	88.72	88.92	89.16	89.42	89.69	89.98	90.27	90.54	90.79	91.01	91.19	91.32	91.42
8	88.13	88.14	88.15	88.18	88.24	88.32	88.43	88.58	88.75	88.95	89.17	89.40	89.64	89.88	90.10	90.31	90.48	90.63	90.74	90.83
9	87.98	87.98	88.00	88.04	88.09	88.17	88.28	88.40	88.55	88.72	88.91	89.10	89.30	89.50	89.68	89.85	89.99	90.11	90.20	90.26
10	87.81	87.81	87.84	87.87	87.93	88.00	88.09	88.21	88.34	88.49	88.64	88.81	88.97	89.13	89.28	89.41	89.53	89.62	89.68	89.72
11	87.63	87.63	87.65	87.69	87.74	87.81	87.90	88.00	88.11	88.24	88.37	88.51	88.64	88.77	88.89	89.00	89.08	89.15	89.19	89.21
12	87.43	87.44	87.46	87.49	87.54	87.61	87.68	87.77	87.87	87.98	88.09	88.21	88.32	88.42	88.52	88.60	88.66	88.70	88.72	88.72
13	87.22	87.23	87.25	87.28	87.33	87.39	87.46	87.54	87.62	87.72	87.82	87.91	88.00	88.09	88.16	88.22	88.26	88.28	88.28	88.25
14	87.01	87.01	87.03	87.06	87.10	87.16	87.22	87.29	87.37	87.46	87.54	87.62	87.70	87.77	87.82	87.86	87.89	87.89	87.86	87.81
15	86.78	86.78	86.80	86.83	86.87	86.92	86.98	87.05	87.12	87.19	87.27	87.34	87.40	87.46	87.50	87.52	87.53	87.51	87.47	87.40
16	86.54	86.54	86.56	86.59	86.63	86.67	86.73	86.79	86.86	86.93	87.00	87.06	87.12	87.16	87.19	87.21	87.20	87.17	87.11	87.01
17	86.28	86.29	86.31	86.34	86.37	86.42	86.48	86.54	86.60	86.67	86.73	86.79	86.84	86.88	86.91	86.91	86.89	86.85	86.77	86.67
18	86.02	86.03	86.05	86.08	86.11	86.16	86.22	86.28	86.35	86.41	86.48	86.53	86.58	86.62	86.64	86.64	86.61	86.56	86.48	86.36
19	85.75	85.76	85.78	85.81	85.85	85.90	85.96	86.02	86.09	86.16	86.22	86.28	86.34	86.37	86.39	86.39	86.36	86.31	86.22	86.09
20	85.47	85.48	85.49	85.53	85.57	85.62	85.69	85.76	85.83	85.91	85.98	86.05	86.10	86.14	86.17	86.17	86.14	86.09	85.99	85.86
21	85.17	85.18	85.20	85.24	85.28	85.34	85.41	85.49	85.57	85.66	85.74	85.82	85.88	85.94	85.97	85.98	85.95	85.90	85.81	85.68
22	84.87	84.87	84.90	84.94	84.99	85.05	85.13	85.22	85.32	85.41	85.51	85.60	85.68	85.75	85.79	85.81	85.80	85.75	85.67	85.54
23	84.54	84.55	84.58	84.62	84.68	84.75	84.84	84.94	85.06	85.17	85.29	85.40	85.50	85.58	85.64	85.68	85.68	85.65	85.57	85.46
24	84.20	84.21	84.24	84.29	84.35	84.44	84.54	84.66	84.79	84.93	85.07	85.21	85.33	85.44	85.52	85.57	85.60	85.58	85.52	85.42
25	83.85	83.86	83.89	83.94	84.01	84.11	84.23	84.36	84.52	84.69	84.86	85.03	85.18	85.32	85.43	85.51	85.55	85.55	85.51	85.42
26	83.47	83.48	83.52	83.57	83.65	83.76	83.89	84.05	84.24	84.44	84.65	84.86	85.05	85.23	85.37	85.48	85.55	85.57	85.55	85.48
27	83.08	83.09	83.12	83.17	83.26	83.37	83.53	83.72	83.94	84.19	84.45	84.71	84.95	85.16	85.34	85.49	85.58	85.64	85.64	85.60
28	82.67	82.68	82.70	82.75	82.84	82.96	83.13	83.35	83.62	83.92	84.25	84.57	84.87	85.13	85.36	85.54	85.67	85.75	85.78	85.76
29	82.24	82.25	82.26	82.30	82.37	82.50	82.68	82.93	83.26	83.64	84.05	84.45	84.82	85.15	85.42	85.64	85.80	85.92	85.97	85.98
30	81.81	81.80	81.79	81.80	81.86	81.98	82.16	82.44	82.83	83.33	83.87	84.37	84.82	85.21	85.54	85.80	85.99	86.13	86.22	86.25
31	81.41	81.37	81.29	81.26	81.30	81.39	81.56	81.84	82.31	83.00	83.71	84.34	84.89	85.35	85.72	86.01	86.24	86.41	86.52	86.57
32	81.08	80.98	80.73	80.67	80.67	80.72	80.83	81.06	81.55	82.64	83.62	84.41	85.05	85.56	85.97	86.30	86.55	86.74	86.88	86.95
33	80.94	80.73	***80.00***	***80.00***	***80.00***	***80.00***	***80.00***	***80.00***	***80.20***	82.40	83.71	84.63	85.33	85.88	86.31	86.66	86.93	87.14	87.29	87.38
34	81.25	80.98	***80.00***	***79.80***	***79.60***	***79.70***	***79.80***	***80.00***	81.57	83.05	84.20	85.07	85.76	86.31	86.74	87.10	87.37	87.59	87.75	87.86

Table B.2: MODFLOW Model - Stage 2 Input Hydraulic Heads

	Column 1	2	3	4	5	6	7	8	9	10	11	12	13	14	15	16	17	18	19	20
35	82.09	81.96	81.60	81.43	81.37	81.46	81.69	82.13	83.04	84.05	84.95	85.70	86.33	86.84	87.26	87.60	87.88	88.10	88.26	88.38
36	83.21	83.15	83.02	82.95	82.97	83.10	83.36	83.79	84.42	85.14	85.84	86.47	87.01	87.47	87.86	88.18	88.44	88.65	88.82	88.95
37	84.43	84.42	84.38	84.39	84.46	84.61	84.86	85.23	85.71	86.26	86.80	87.32	87.78	88.18	88.52	88.82	89.06	89.26	89.42	89.54
38	85.69	85.70	85.71	85.76	85.86	86.02	86.26	86.57	86.95	87.37	87.80	88.22	88.60	88.94	89.24	89.50	89.72	89.90	90.05	90.17
39	86.95	86.96	87.00	87.08	87.20	87.36	87.57	87.83	88.14	88.47	88.81	89.15	89.46	89.75	90.00	90.22	90.42	90.58	90.71	90.83
40	88.18	88.20	88.26	88.36	88.49	88.65	88.84	89.06	89.30	89.57	89.84	90.10	90.35	90.58	90.79	90.98	91.14	91.28	91.40	91.50
41	89.38	89.41	89.49	89.60	89.75	89.90	90.07	90.25	90.45	90.66	90.86	91.07	91.26	91.44	91.61	91.76	91.89	92.00	92.10	92.19
42	90.51	90.56	90.67	90.83	90.99	91.14	91.29	91.44	91.59	91.74	91.89	92.04	92.19	92.32	92.44	92.56	92.65	92.74	92.82	92.89
43	91.56	91.63	91.82	92.04	92.24	92.39	92.51	92.62	92.72	92.83	92.93	93.03	93.12	93.21	93.29	93.36	93.43	93.49	93.54	93.59
44	92.46	92.61	92.92	93.27	93.56	93.67	93.75	93.81	93.86	93.91	93.96	94.01	94.06	94.10	94.14	94.18	94.21	94.24	94.27	94.30
45	93.07	93.41	93.97	94.57	95.03	***95.00***	***95.00***	***95.00***	***95.00***	***95.00***	***95.00***	***95.00***	***95.00***	***95.00***	***95.00***	***95.00***	***95.00***	***95.00***	***95.00***	***95.00***
46	***93.00***	***94.00***	***95.00***	***96.00***	***97.00***	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

Table B.2: MODFLOW Model - Stage 2 Input Hydraulic Heads (continued)

	Column 21	22	23	24	25	26	27	28	29	30	31	32	33	34	35	36	37	38	39
Row 1	***96.00***	***96.00***	***96.00***	***96.00***	***96.00***	***95.00***	***94.00***	***93.00***	***92.00***	***92.00***	***93.00***	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
2	95.14	95.14	95.11	95.03	94.81	94.26	93.61	92.97	92.47	92.39	92.69	***93.00***	0.00	0.00	0.00	0.00	0.00	0.00	0.00
3	94.31	94.32	94.28	94.18	93.97	93.62	93.21	92.81	92.52	92.40	92.36	***92.00***	0.00	0.00	0.00	0.00	0.00	0.00	0.00
4	93.53	93.55	93.52	93.43	93.27	93.04	92.79	92.55	92.39	92.33	92.37	92.49	***93.00***	0.00	0.00	0.00	0.00	0.00	0.00
5	92.80	92.83	92.81	92.74	92.63	92.49	92.35	92.22	92.15	92.17	92.29	92.60	***93.50***	0.00	0.00	0.00	0.00	0.00	0.00
6	92.12	92.15	92.14	92.10	92.04	91.96	91.88	91.83	91.83	91.90	92.01	92.13	***92.00***	0.00	0.00	0.00	0.00	0.00	0.00
7	91.48	91.51	91.51	91.49	91.45	91.41	91.39	91.40	91.46	91.57	91.74	91.91	***92.00***	0.00	0.00	0.00	0.00	0.00	0.00
8	90.88	90.90	90.90	90.88	90.86	90.85	90.86	90.91	91.02	91.19	91.45	91.78	92.23	***93.00***	0.00	0.00	0.00	0.00	0.00
9	90.30	90.31	90.30	90.29	90.27	90.27	90.30	90.37	90.51	90.74	91.08	91.53	92.14	***93.00***	0.00	0.00	0.00	0.00	0.00
10	89.74	89.74	89.72	89.69	89.67	89.66	89.68	89.76	89.92	90.19	90.58	91.13	91.81	***92.60***	0.00	0.00	0.00	0.00	0.00
11	89.21	89.18	89.14	89.10	89.05	89.02	89.02	89.07	89.22	89.50	89.95	90.58	91.38	***92.30***	0.00	0.00	0.00	0.00	0.00
12	88.69	88.64	88.58	88.50	88.41	88.34	88.29	88.29	88.39	88.65	89.12	89.86	90.84	***92.00***	0.00	0.00	0.00	0.00	0.00
13	88.20	88.12	88.02	87.91	87.77	87.64	87.51	87.42	87.41	87.57	88.03	88.90	90.11	91.52	***93.00***	0.00	0.00	0.00	0.00
14	87.73	87.63	87.49	87.33	87.14	86.92	86.69	86.46	86.27	86.21	86.52	87.62	89.17	90.99	***93.00***	0.00	0.00	0.00	0.00
15	87.29	87.16	86.99	86.78	86.53	86.23	85.88	85.47	84.98	84.47	84.22	85.87	87.97	90.25	***92.60***	0.00	0.00	0.00	0.00
16	86.89	86.72	86.52	86.27	85.96	85.59	85.13	84.54	83.73	82.48	***80.00***	83.68	86.61	89.42	***92.30***	0.00	0.00	0.00	0.00
17	86.52	86.33	86.10	85.81	85.46	85.03	84.50	83.83	82.94	81.71	***80.00***	82.26	85.35	88.54	***92.00***	0.00	0.00	0.00	0.00
18	86.20	85.99	85.74	85.42	85.04	84.58	84.02	83.33	82.48	81.43	***80.20***	***80.00***	83.99	87.39	90.51	***93.00***	0.00	0.00	0.00
19	85.92	85.70	85.43	85.10	84.70	84.23	83.66	82.99	82.21	81.33	80.44	***79.80***	83.22	86.51	89.66	***92.60***	0.00	0.00	0.00
20	85.69	85.46	85.18	84.85	84.44	83.96	83.41	82.77	82.04	81.25	80.43	***79.60***	82.59	85.76	89.01	***92.30***	0.00	0.00	0.00
21	85.50	85.28	85.00	84.66	84.26	83.78	83.24	82.62	81.94	81.21	80.43	***79.60***	81.78	84.93	88.31	***92.00***	0.00	0.00	0.00
22	85.37	85.15	84.87	84.54	84.14	83.67	83.14	82.55	81.90	81.20	80.48	***79.80***	***80.00***	83.89	87.30	90.47	***93.00***	0.00	0.00
23	85.29	85.08	84.81	84.48	84.09	83.63	83.11	82.53	81.89	81.21	80.51	***79.80***	***79.90***	83.32	86.53	89.58	***92.60***	0.00	0.00
24	85.26	85.06	84.80	84.48	84.10	83.65	83.14	82.57	81.93	81.25	80.53	***79.80***	***79.90***	82.96	85.92	88.74	91.38	***93.60***	0.00
25	85.29	85.10	84.85	84.54	84.17	83.74	83.23	82.66	82.02	81.32	80.57	***79.80***	***79.90***	82.70	85.44	88.07	90.57	***93.00***	0.00
26	85.37	85.19	84.96	84.67	84.32	83.89	83.40	82.83	82.17	81.44	80.64	***79.80***	***79.80***	82.49	85.08	87.53	89.82	***92.00***	0.00
27	85.50	85.35	85.14	84.86	84.53	84.12	83.64	83.07	82.40	81.63	80.76	***79.80***	***79.80***	82.38	84.85	87.14	89.20	90.95	***92.00***
28	85.69	85.56	85.37	85.12	84.81	84.43	83.96	83.41	82.74	81.93	80.95	***79.80***	***79.80***	82.40	84.80	86.97	88.91	90.59	***92.00***
29	85.93	85.82	85.66	85.45	85.17	84.81	84.39	83.86	83.21	82.39	81.32	***79.90***	***79.90***	82.62	84.98	87.04	88.86	90.50	***92.00***
30	86.23	86.15	86.02	85.83	85.59	85.28	84.90	84.44	83.86	83.10	82.04	***80.20***	***80.00***	83.20	85.46	87.34	89.01	90.54	***92.00***
31	86.58	86.53	86.43	86.28	86.08	85.82	85.51	85.13	84.67	84.13	83.52	82.94	83.21	84.71	86.32	87.85	89.29	90.67	***92.00***
32	86.98	86.96	86.89	86.78	86.62	86.42	86.17	85.89	85.57	85.24	84.96	84.85	85.20	86.11	87.25	88.45	89.65	90.83	***92.00***
33	87.43	87.43	87.39	87.32	87.20	87.06	86.88	86.69	86.49	86.31	86.22	86.28	86.63	87.28	88.12	89.05	90.03	91.01	***92.00***
34	87.93	87.95	87.94	87.90	87.82	87.72	87.61	87.49	87.38	87.31	87.31	87.45	87.76	88.25	88.89	89.62	90.39	91.19	***92.00***

Table B.2: MODFLOW Model - Stage 2 Input Hydraulic Heads (continued)

	Column 21	22	23	24	25	26	27	28	29	30	31	32	33	34	35	36	37	38	39
35	88.46	88.51	88.52	88.50	88.47	88.41	88.34	88.28	88.23	88.22	88.28	88.43	88.70	89.08	89.57	90.13	90.73	91.36	***92.00***
36	89.04	89.09	89.13	89.13	89.13	89.10	89.08	89.06	89.05	89.08	89.15	89.30	89.52	89.82	90.18	90.60	91.05	91.52	***92.00***
37	89.64	89.71	89.76	89.78	89.80	89.80	89.81	89.81	89.83	89.88	89.96	90.09	90.26	90.48	90.75	91.04	91.35	91.67	***92.00***
38	90.27	90.35	90.40	90.45	90.48	90.51	90.53	90.56	90.59	90.65	90.72	90.83	90.96	91.11	91.28	91.45	91.63	91.81	***92.00***
39	90.92	91.00	91.07	91.12	91.17	91.21	91.25	91.29	91.34	91.39	91.46	91.54	91.63	91.73	91.81	91.87	91.91	91.95	***92.00***
40	91.59	91.67	91.74	91.80	91.87	91.92	91.98	92.02	92.07	92.12	92.18	92.24	92.30	92.35	92.36	92.30	92.20	92.09	***92.00***
41	92.27	92.34	92.41	92.49	92.56	92.64	92.70	92.75	92.80	92.84	92.89	92.94	92.99	93.02	92.98	92.77	92.48	92.20	***92.00***
42	92.95	93.02	93.09	93.17	93.27	93.37	93.44	93.49	93.53	93.56	93.60	93.64	93.69	93.75	93.76	93.32	92.76	92.24	***92.00***
43	93.64	93.69	93.75	93.83	93.96	94.13	94.20	94.24	94.26	94.28	94.30	94.33	94.39	94.54	***95.00***	***94.00***	***93.00***	***92.00***	***92.00***
44	94.32	94.35	94.39	94.46	94.60	***95.00***	***95.00***	***95.00***	***95.00***	***95.00***	***95.00***	***95.00***	***95.00***	***95.00***	0.00	0.00	0.00	0.00	0.00
45	***95.00***	***95.00***	***95.00***	***95.00***	***95.00***	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
46	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00

APPENDIX C: Stage 1 Modelling Input Files

Table C.1: Input ASCII Files for MT3D Model - Stage 1

SBT1BTN.INP

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CASE STUDY, ORIGINAL FLOW FIELD TRANSPORT ANALYSIS
SOUTH BRUNSWICK TOWNSHIP, NJ
         1        46        39         1
 DAY  FT  LB
 T T T T
 1
      103         0                   -1
5*200.  4*100. 16*50.  4*100. 10*200.
      103         0                   -1
4*100.  16*50.   8*100.  18*200.
         0       95.
         0       35.
         0       .40
         1         1(39I3)            1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  0  0  0  0  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  0  0  0  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  0  0  0  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  0  0  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  0  0  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  0  0  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  0  0  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  0  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  0  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1 -1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  0  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  0  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  0  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1  1
```

```
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  0  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  0  0  0
0  0  0  0  0  0  0  0  0  0  0
  1  1  1  1  1  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0
0  0  0  0  0  0  0  0  0  0  0
       101         0                            -1
         2
  1 46  1 39 0.
 10 10 18 18  40000.
      -99.
        -1         0         0         0         T
         0
         0
         T
      36500       10         1.
         0     50000
```

SBT1ADV.INP

```
         2       .75     10000
         2       .75
         1         0        16
```

SBT1DSP.INP

```
         0        3.
         0        .1
         0       .01
         0        0.
```

SBT1SSM.INP

```
 T F T F F F
      1000
         1
         0        0.
         1
```

```
     1          10          18       40000          2
```

SBT1RCT.INP

```
     1           0
     0        103.
     0        .003
     0          0.
```

APPENDIX D: Stage 3 Modelling Input Files

Table D.1: Input ASCII Files for MODFLOW and MT3D Models - Stage 3

MODFLOW:

SBT3BAS.DAT

```
CASE STUDY, TRANSIENT, PUMP-AND-TREAT OPERATION ANALYSIS
SOUTH BRUNSWICK TOWNSHIP
         1          46          39          1          4
 11 12 00  0  0  0  0 18 19  0  0 22 00 00 00 00 00 00 00 00 00 23 00 00
         1           0
         1           1(3913)                           1
    -1  -1  -1  -1  -1  -1  -1  -1  -1  -1  -1  -1  -1  -1  -1  -1  -1  -1  -1  -1  -1  -1  -1  -1  -1  -1  -1  -1
  -1  -1  -1   0   0   0   0   0   0   0   0
     1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1  -1  -1   1  -1  -1  -1   1   1   1   1
   1   1   1  -1   0   0   0   0   0   0   0
     1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1  -1   1   1   1   1   1   1   1
   1   1   1  -1   0   0   0   0   0   0   0
     1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
   1   1   1   1  -1   0   0   0   0   0   0
     1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
   1   1   1   1  -1   0   0   0   0   0   0
     1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
   1   1   1   1  -1   0   0   0   0   0   0
     1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
   1   1   1   1  -1   0   0   0   0   0   0
     1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
   1   1   1   1   1  -1   0   0   0   0   0
     1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
   1   1   1   1   1  -1   0   0   0   0   0
     1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
   1   1   1   1   1  -1   0   0   0   0   0
     1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
   1   1   1   1   1  -1   0   0   0   0   0
     1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
   1   1   1   1   1  -1   0   0   0   0   0
     1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
   1   1   1   1   1   1  -1   0   0   0   0
     1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
   1   1   1   1   1   1  -1   0   0   0   0
     1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
   1   1   1   1   1   1  -1   0   0   0   0
     1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
   1   1  -1   1   1   1  -1   0   0   0   0
     1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
   1   1  -1   1   1   1  -1   0   0   0   0
     1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
   1   1  -1  -1   1   1   1  -1   0   0   0
     1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
   1   1   1  -1   1   1   1  -1   0   0   0
     1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
   1   1   1  -1   1   1   1  -1   0   0   0
     1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
   1   1   1  -1   1   1   1  -1   0   0   0
     1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
   1   1   1  -1  -1   1   1      -1   0   0
     1   1   1   1   1   1                   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
   1   1   1  -1  -1   1   1               0
     1   1   1   1   1   1                   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
   1   1   1  -1  -1   1   1
     1   1   1   1   1   1                       1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
   1   1   1  -1  -1   1   1
     1   1   1   1   1   1   1                   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
   1   1   1  -1  -1   1   1
     1   1   1   1   1   1   1                   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
   1   1   1  -1  -1   1   1   1
     1   1   1   1   1   1   1   1           1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
   1   1   1  -1  -1   1   1   1   1   1  -1
     1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
   1   1   1  -1  -1   1   1   1   1   1  -1
     1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
   1   1   1  -1  -1   1   1   1   1   1  -1
     1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
   1   1   1   1   1   1   1   1   1   1  -1
     1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
   1   1   1   1   1   1   1   1   1   1  -1
```

```
   1   1  -1  -1  -1  -1  -1  -1  -1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
1   1   1   1   1   1   1   1   1   1  -1
   1   1  -1  -1  -1  -1  -1  -1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
1   1   1   1   1   1   1   1   1   1  -1
   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
1   1   1   1   1   1   1   1   1   1  -1
   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
1   1   1   1   1   1   1   1   1   1  -1
   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
1   1   1   1   1   1   1   1   1   1  -1
   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
1   1   1   1   1   1   1   1   1   1  -1
   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
1   1   1   1   1   1   1   1   1   1  -1
   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
1   1   1   1   1   1   1   1   1   1  -1
   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
1   1   1   1   1   1   1   1   1   1  -1
   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
1   1   1   1   1   1   1   1   1   1  -1
   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
1   1   1   1   1   1  -1  -1  -1  -1  -1
   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1   1
-1  -1  -1  -1  -1  -1   0   0   0   0   0
   1   1   1   1   1  -1  -1  -1   1  -1  -1  -1  -1  -1  -1  -1  -1  -1  -1  -1  -1  -1  -1  -1   1   0   0   0
0   0   0   0   0   0   0   0   0   0   0
  -1  -1  -1  -1  -1   0   0   0   0   0   0   0   0   0   0   0   0   0   0   0   0   0       0   0   0   0   0
0   0   0   0   0   0   0   0   0   0   0
        -99.
         -31          1.
         300          (
```

SBT3BCF.DAT

```
           0          31
           1
           0          1.
          11          0.(14F6.0)                        -1
 200.    200.    200.    200.    200.    100.    100.    100.    100.     50.     50.     50.     50.     50.
  50.     50.     50.     50.     50.     50.     50.     50.     50.     50.     50.    100.    100.    100.
 100.    200.    200.    200.    200.    200.    200.    200.    200.    200.    200.
          11          0.(16F6.0)                        -1
 100.    100.    100.    100.     50.     50.     50.     50.     50.     50.     50.     50.     50.     50.
50.     50.
  50.     50.     50.     50.    100.    100.    100.    100.    100.    100.    100.    100.    200.    200.
200.    200.
 200.    200.    200.    200.    200.    200.    200.    200.    200.    200.    200.    200.    200.    200.
         101          0                             -1
2
  1   5   1  39  150.
  6  46   1  39  350.
           0         60.
```

SBT3WEL.DAT

```
           7          31
           7
           1           8          17       -3780
           1          11          12       -3780
           1          10          18       -3780
           1          10          20       -3780
           1          12          18       -3780
           1          13          15       -3780
           1          16          17       -3780
```

MT3D:

SBT3BTN.INP

```
CASE STUDY, PUMP-AND-TREAT OPERATION ANALYSIS
SOUTH BRUNSWICK TOWNSHIP, NJ
         1        46        39         1
 DAY  FT  LB
 T T T T
 1
       103         0                             -1
 5*200.  4*100. 16*50.  4*100. 10*200.
       103         0                             -1
 4*100.  16*50.   8*100.  18*200.
         0       95.
         0       35.
         0       .40
         1        1(39I3)                        1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  0  0  0  0  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  0  0  0  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  0  0  0  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  0  0  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  0  0  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  0  0  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  0  0  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  0  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  0  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  0  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  0  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  0  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  0  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  0  0  0  0
  1  1  1  1  1  1  1  1  1  1     1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1  1
```

```
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  1  1  1  1  1
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1
1  1  1  1  1  1  0  0  0  0  0
  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  1  0  0  0
0  0  0  0  0  0  0  0  0  0  0
  1  1  1  1  1  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0  0
0  0  0  0  0  0  0  0  0  0  0
        -41        1.
       -99.
         -1         0          0          0          T
          0
          0
          T
        300        10         1.
          0     50000
```

SBT3SSM.INP

```
 T F T F F F
       1000
          1
          0         0.
          0
```