GROUNDWATER MONITORING IN THREE DIMENSIONS USING THE MULTI-LEVEL SAMPLER

by

Jennifer A. Hyman

SUBMITTED TO THE DEPARTMENT OF CIVIL
ENGINEERING IN PARTIAL FULFILLMENT OF THE
REQUIREMENTS FOR THE DEGREE OF

MASTER OF SCIENCE

at the

MASSACHUSETTS INSTITUTE OF TECHNOLOGY

June 1990

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Submitted to the Department of Civil Engineering on June 3, 1990 in partial fulfillment of the requirements for the degree of Master of Science.

Abstract

A groundwater monitoring program was carried out at a coal tar-contaminated site known as Site 24. In order to better describe the naphthalene plume in three dimensions, experimental multi-level samplers (MLSs) were incorporated into the conventional groundwater monitoring program.

A unique MLS design, made from materials selected for their inertness, was developed to sample for the organic contamination. A number of criteria were considered in establishing this design, including cost, contaminant type, and installation and sampling techniques available. The design was based on MLSs sold commercially and used extensively in three large tracer tests studies. The most common critique of MLSs is that incomplete caving of the aquifer around the MLS will allow groundwater to be sampled from areas outside of each sampling port zone. In situ tracer test studies carried out on our MLSs and at these three research sites suggest that in general, this problem of crosstalk between sampling ports does not exist.

The data generated from the fourteen MLSs installed give much better resolution of the plume in the vertical than the monitoring well clusters installed at Site 24. Monitoring wells prove to be more efficient at delineating the horizontal boundaries of the plume. The groundwater data from monitoring wells and MLSs do not correlate well with data from soil analyses. Variation of the MLS data over time could not be explained by seasonal recharge events. The combination of MLS and monitoring well data provides a well-resolved three-dimensional picture of the naphthalene plume that is narrow and long, travels from the source to seeps, and seems to start above a clay lens in the aquifer, but end up entirely below the lens. The special installation of a MLS to sample above and below the clay lens demonstrates the usefulness of the MLS to detect complicated distributions of contaminants in the vertical. In sum, the MLSs designed for Site 24 are well-suited for detecting vertical distributions of semivolatile organic contaminations in fairly sandy, relatively homogeneous aquifers.

Thesis Supervisor:	Professor Dennis McLaughlin
Title:	Professor of Civil Engineering

Dedication

I would like to thank a number of people for helping to make the dream of receiving an advanced degree from MIT a reality. First, I must thank Dennis, my advisor, for encouraging me in my work and allowing me the time and the resources to be an MIT graduate student and still take part in extra-curricular activities. Secondly, I would like to thank Dr. Ishwar Murarka of the Electric Power Research Institute for having the faith in me as an engineer to fund the second and most decisive portion of my research. Thirdly, I extend many yards of gratitude to my friend and co-worker Lynn B. Reid. If it weren't for her strong back and warm heart, I might not have made it through some very tough times in this project. My last round of thanks goes out to my family and friends, who with their love, have taught me to believe in myself and set my sights high.

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Chapter 1

Background

1.1 The Importance of Groundwater

The water that occurs beneath the water table in soils and geologic formations that are fully saturated is called groundwater (Freeze and Cherry, 1979). Groundwater is a vast and valuable resource. The U.S. Geological Survey National Water Summary of 1988 reported that groundwater is the source of drinking water for more than half of the U.S. population. In addition, on average, 40 percent of the annual stream flow in the U.S. is comprised of groundwater seepage. Because of our dependence on clean groundwater for domestic, agricultural and recreational uses, contamination of these sources is an important topic to investigate. The more contaminated aquifers are scientifically examined, the more knowledge we will gain about the behavior of contaminants in groundwater and as a result, groundwater data can be more efficiently taken.

1.2 Traditional Groundwater Monitoring Techniques

In order to determine the extent of present and future contamination in the subsurface, it is necessary to perform groundwater monitoring. For assessments of uncontrolled sites, groundwater monitoring primarily involves locating the source of the contamination and the extent to which the contaminants have been transported away from the source. From this information on where the different levels of contamination are, risk assessments and remedial investigations may be performed. In order to try to fully understand contaminant transport in a site, it is necessary to determine hydrologic properties of the aquifer, including water table elevations, hydraulic conductivity and porosity. Subsurface characteristics are often poorly correlated in space depending on the degree of soil layering and natural heterogeniety. In order to optimally assess the extent of groundwater pollution at any point in time (and be able to predict where it will be in the future) it is important to have information on the concentrations of the chemicals of interest in the three spatial dimensions of the site. It is desirable, therefore, to gather accurate data in as many of these dimensions as is economically practicable. With all the spatial and temporal variations of real sites, it is a challenge to the environmental engineer to collect groundwater data that are reliable and interpretable.

1.2.1 The Monitoring Well

The conventional way of accessing the groundwater for the determination of water quality and the measurement of water levels is through monitoring wells installed in the aquifer. A monitoring well is a narrow pipe that is slotted or screened over a number of feet at the closed bottom end of a pipe. The groundwater enters the well from this screened area only. Groundwater samples taken from a a monitoring well are a mixture or an average of the groundwater that enters from the screened length of the monitoring well. Monitoring wells can either partially or fully penetrate the aquifer, and they are traditionally made of stainless steel, PVC, or Teflon. A diagram of a monitoring well partially penetrating an aquifer is shown in Figure 1-1. This figure shows the mixing of uncontaminated and contaminated groundwater that can occur in a monitoring well.

The material selected for use in construction of a monitoring well depends on the application. While cost may be a factor, the over-riding influence on material selection is the maintenance of sample integrity. Materials should be specified which limit the possibility of leaching chemical constituents into the water sample, or adsorbing the contaminants from the water sample onto the monitoring well itself. This is an important issue which will be taken up in greater detail later on in this thesis.

A wide variety of techniques are available to install monitoring wells. Most of these

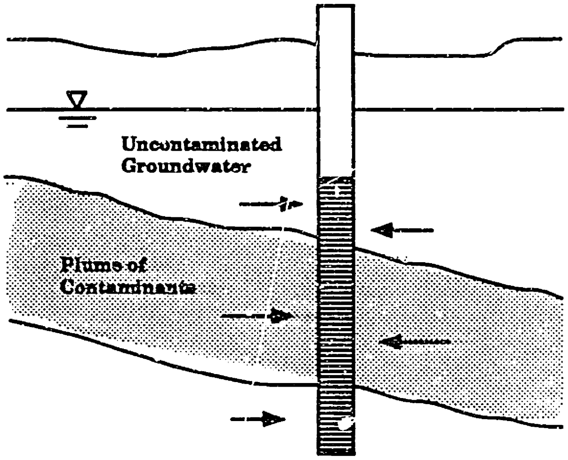


Figure 1-1: Mixing of contaminated and uncontaminated groundwater in a monitoring well partially penetrating an aquifer.

methods require the use of a drilling rig. Once a hole is drilled and the monitoring well is placed in it, sand is usually packed around the screen to prevent clogging of the screen with fines (soil particles of the smallest particle size fraction) from the aquifer. The remaining annular space between the well and the borehole wall is grouted to assure that the bore hole is less permeable than the surrounding soil. A recommended grout is a mixture of cement and bentonite, a material which swells on contact with water. A well casing is cemented over the top of the monitoring well at the ground surface for protection and to prevent rainwater from streaming down into the aquifer along the outside of the well. The last part of the installation process is called well development. Monitoring wells are developed by either pumping water into or out of the well to dislodge or remove small soil particles introduced in drilling. This is done in order to avoid future clogging of the well screen and to initiate the process of re-establishing equilibrium chemical and hydrological conditions around the well. Proper technique and duration of well development have not yet been scientifically established.

1.2.2 The Piezometer

A piezometer is basically a cheap monitoring well, whose main purpose in groundwater monitoring is to measure the hydraulic head at a particular point in an aquifer. It is useful to know the hydraulic head at many points in an aquifer because from the head gradients the direction and magnitude of groundwater velocites can be calculated. The direction of groundwater flow is perpendicular to lines of constant head and groundwater velocity is proportional to the size of the gradient. Where and how fast the groundwater is moving can be a good predictor of the size and shape of the contamination plume.

The hydraulic head at some point in an aquifer, $h(x,y,z)$, is the sum of the pressure head and the elevation head:

$$h(x,y,z) = p(x,y,z)/\rho g + z$$

where $p(x,y,z)$ is the pressure at some point in the aquifer, ρ is the density of the water, g is the acceleration due to gravity and z is the distance of the point above a $z = 0$ datum. Figure 1-2 shows the relationship between the hydraulic head, pressure head and elevation head for a piezometer (adapted from Freeze and Cherry, 1979). Pressure at a point (x,y,z) is defined by:

$$p(x,y,z) = \rho g \psi + p_0$$

where ψ is the height of the water column above the point and p_0 is atmospheric pressure. Since atmospheric pressure is zero by convention, the head expression can be written:

$$h = \psi + z .$$

The hydraulic head at any point in an aquifer becomes the elevation of the height of the water in the piezometer over sea level.

Piezometers are commonly made of PVC plumbing pipe, with a slotted, pointed section at the bottom. Shallow installation of piezometers may be simply accomplished

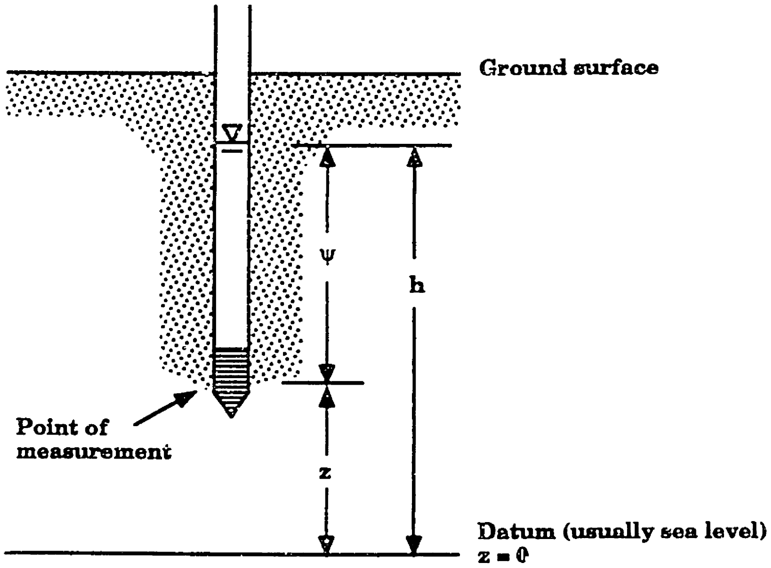


Figure 1-2: Hydraulic head (h), pressure head (ψ), and elevation head (z) for a piezometer in an unconfined aquifer.

with a hand auger, while deeper installation may require the use of a drill rig. In order to take a water level measurement from a piezometer, the top of each piezometer casing is surveyed. Water table elevations are found by subtracting the unwetted length of piezometer from the casing top elevation.

Clusters of piezometers or monitoring wells screened at different depths within the aquifer may be used to examine the presence of vertical head gradients. Significant vertical head gradients can be responsible for vertical shifts in contaminant transport and thus are important to consider as part of a groundwater monitoring program. A comparison of the vertical and horizontal head gradients can give some indication of the significance of vertical flow.

1.3 Groundwater Monitoring Networks

Because of the individuality of each site there is no common recipe for designing groundwater monitoring networks. Groundwater engineers are not at liberty to prescribe any number of monitoring wells for a site assessment because of the extremely high cost of drilling time and chemical analyses, and often also due to EPA regulations that specify where and how many monitoring wells to install. The few monitoring wells that can be installed must therefore be carefully sited where they will collect the most useful data.

In an ideal world that is everywhere homogeneous and isotropic, chemicals spread out from a source of contamination in the groundwater in a contiguous symmetric manner, forming an elliptical plume, shown in Figure 1-3. The shape of a plume is dependent on the groundwater velocity, the degree to which the chemicals disperse, the degree to which the chemicals' transport is retarded by adsorption, chemical reaction or biological degradation, and the hydraulic conductivities of the soils throughout the aquifer. Due to the wide variation of soil properties from point to point, real plumes have very erratic shapes, and can even finger or branch. In the vertical dimension, plumes can dip and sink due do recharge or high specific gravity.

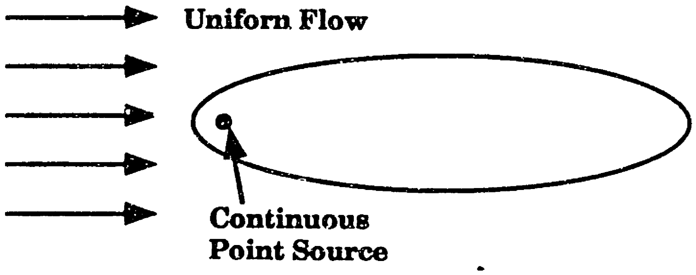


Figure 1-3: The shape of an ideal plume with a constant source, in an isotropic aquifer, with a uniform flow field, at a particular point in time.

The combination of hydrodynamic dispersion resulting from molecular diffusion and advection or mechanical dispersion of the chemicals, determines whether the plume will be short and fat or long and skinny. In a typical site with a substantial head gradient, advection will cause more spreading in the longitudinal direction (long) than in directions normal to the flow axis (skinny). In such a situation, the concentration gradient is higher in the transverse direction than in the longitudinal direction.

A technique that utilizes an assessment of the directional concentration gradient involves placing monitoring wells closer together along the axis where the concentration gradient is highest. From this method, monitoring wells along transects of the plume can provide detail of cross-sections of the plume. In statistical terms, the contaminant concentration is less correlated where the gradient is highest (Graham and McLaughlin, 1989a,b). It is not necessary to place monitoring wells as close together where the concentration is better correlated. Generally, this technique prescribes a grid of monitoring wells or stations arranged on the site. The spacing of the grid will be dependent on the correlation of the aquifer variables, the variability at the site and the acceptable uncenainty (Andelman and Underhill, 1987).

Figures 1-4 and 1-5 show hypothetical groundwater monitoring networks based on the directional correlation of the chemical concentration in the plumes. In the first figure advection dominates, concentration is less correlated and the concentration gradient highest in the transverse direction, and so transverse rows of monitoring wells are set up to gather data where the least is known. In the second figure the head gradients are very small and transverse dispersion dominates. The concentration gradients are highest parallel to the direction of flow so the monitoring wells in the grid are closer together in this direction.

Graham and McLaughlin describe a method for estimating the concentration uncertainty over a site based on available head and concentration data. This method may be used to design a sequential sampling program. An initial (naive) uncertainty map is used to locate the first set of monitoring wells. Data from these wells are used to update the

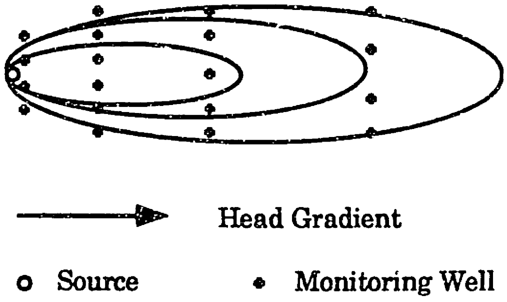


Figure 1-4: A groundwater monitoring network grid on a site with strong head gradients and high correlation of concentration in the direction of flow.

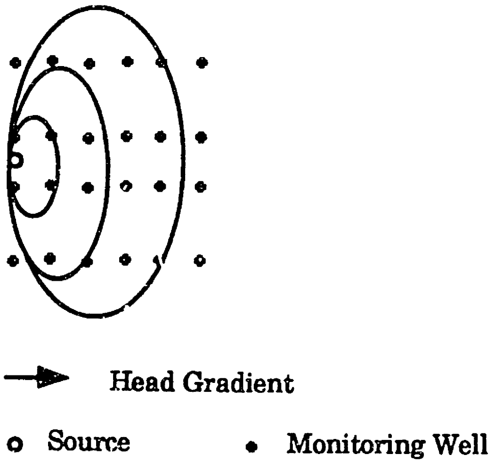


Figure 1-5: A groundwater monitoring network grid on a site with small head gradients (dispersion dominant) and high correlation of concentration normal to the direction of flow.

uncertainty map which is, in turn, used to design another round of sampling. The sampling rounds continue until the uncertainty at the site has been lowered to an acceptable level.

Another strategy of groundwater monitoring is to design the sampling network to find the edges of the plume. In this technique, monitoring wells are specifically located where no contamination is expected, to verify where the plume is not. Figure 1-6 gives an example of what such a monitoring network could look like. Groundwater monitoring networks are based on what they are ultimately trying to achieve, whether it's one well monitoring for leakage down-gradient of a septic tank to hundreds of wells in a government-funded large-scale tracer test study.

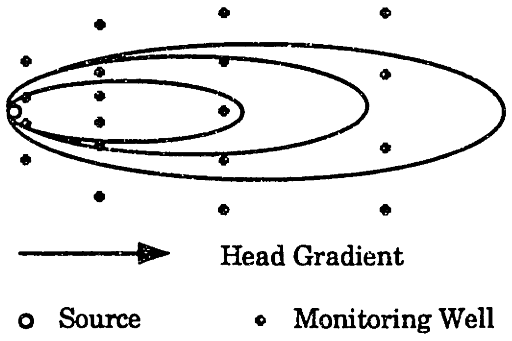


Figure 1-6: A groundwater monitoring network set up to find the edges of the plume.

1.4 The Purpose of this Study: Groundwater Monitoring in Three Dimensions

1.4.1 Project Description

This thesis describes a groundwater monitoring program that involves the collection of three-dimensional concentration data along with traditional depth-averaged data from monitoring wells. Multi-level sampling (MLS) devices, which collect a number of groundwater samples from discrete locations in the vertical from a single well, were constructed and installed at a coal-tar contaminated site. Cambridge Analytical Associates (currently NET Environmental from Bedford, MA), a private environmental consulting firm, was responsible for monitoring well siting, sampling and analysis. Atlantic Environmental Services (AES of Colchester, CT), a hydrogeological contractor, supervised the drilling and collection of head data and soil profiles. The work of these two companies provided the backdrop of typical groundwater and hydrogeological monitoring with which to compare our MLS data.

1.4.2 Specific Goals

There are a number of important aspects of this groundwater monitoring project that I chose to look at in depth. Unique to my effort is the MLS design, developed specifically for determining accurate concentrations of organic contaminants in groundwater. I will go over the issues involving the selection of sampling materials, various MLS designs and installation techniques that have been examined in other programs, and the design and mode of installation that were utilized especially for our site.

Currently, except for a few specific applications, the multi-level sampler is not a familiar or trusted method of taking representative groundwater samples. I will address the most common complaint associated with this technique- the issue of aquifer disturbance during installation affecting the authenticity of the water samples taken (Vonhof and Whitaker, 1979). Small-scale tracer tests were performed to examine this issue and will be described. This thesis will also discuss the relatively innovative idea of integrating MLS use in a conventional groundwater monitoring network. Finally, an attempt will be made to compare the MLS data with the monitoring well data and soil data, with the perspective of the soil stratifications in the aquifer (interpreted from soil borings). Data analysis will be done to determine if more resolution in the distribution of contaminants can be gained using the MLS without sacrificing accuracy or significant amounts of time and money. At the end of this thesis, I will conclude on the optimal application of the multi-level sampler and recommend areas of future study that can improve the efficiency and accuracy of estimating of the spatial extent of groundwater contamination.

Chapter 2

Site 24 Project Overview

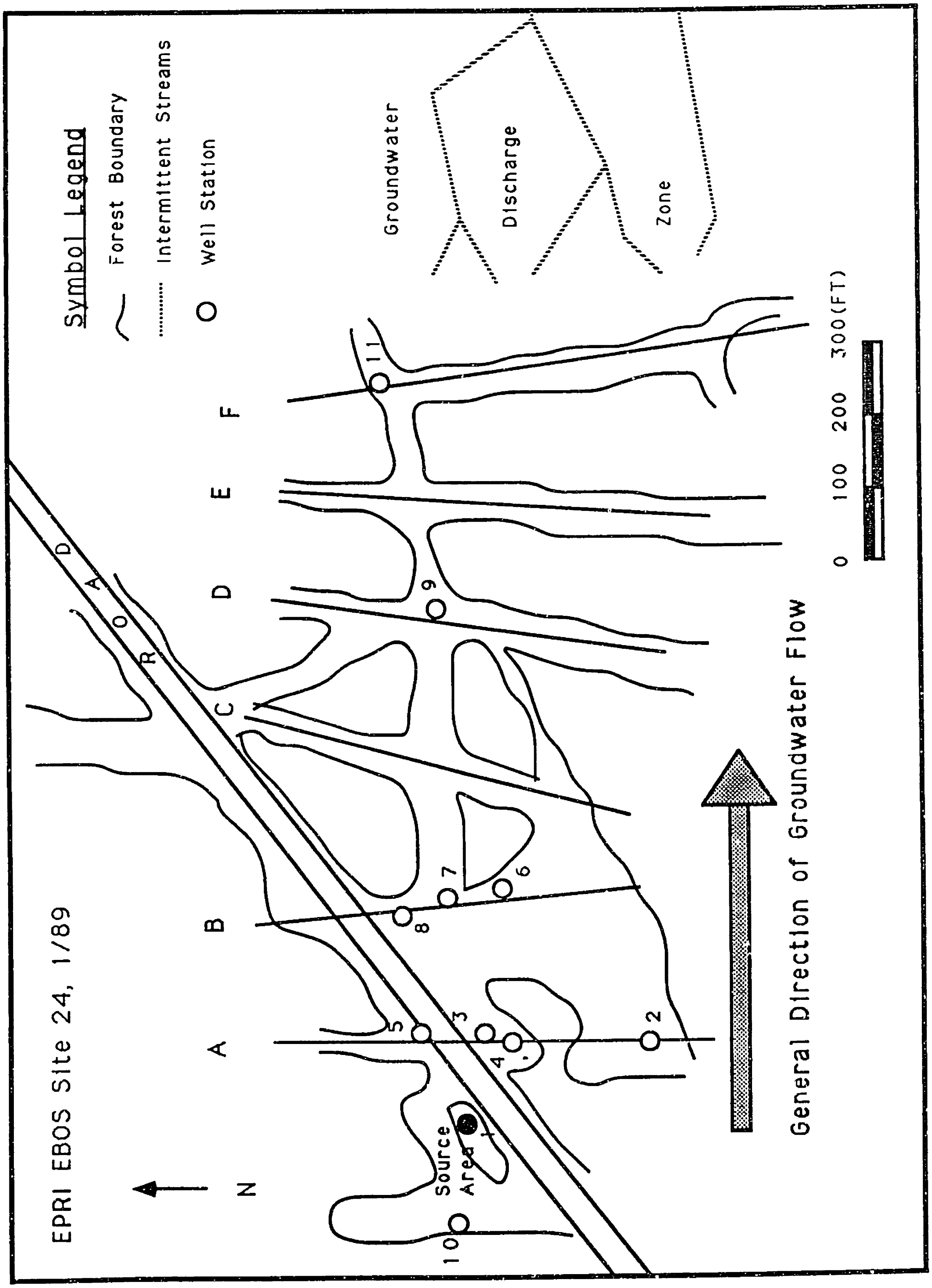
2.1 The Site

This project to examine the role of three-dimensional data in a groundwater monitoring program was conducted at a privately owned contaminated site. This particular site was selected by our group due to its proximity to MIT and the convenience of working with contractors from the Boston area. This site is one of many utility sites contaminated by coal tar residues which is being studied with the direction and funding of the Electric Power Research Institute (EPRI). Our site, EPRI Site 24, is wooded, with one road crossing it, and slopes down towards seeps (streams) which eventually feed into a river. Records show that around 1962, coal tar was dumped into pits by the side of the road. Since then, the soluable components of the coal tar have leached into the groundwater. These organic contaminants have formed a plume which has moved out away from the source in the direction of the seeps. A map of Site 24 showing the surface features on the site when our work began, the approximate location of the original coal tar pits (the source) and the general direction groundwater flow is shown in Figure 2-1.

2.2 The Contamination

Coal tar is a by-product of the the process of coal gasification, a process used in the late 1800s to the period following World War II when the popularity of petroleum brought about its demise (Enzminger and Albert, 1987). During the process coal is heated to drive off or crack organic materials (in the presence of steam in some cases). The gas is then cooled to remove the condensible fraction which is the tar. Before it can be used the gas is washed with iron oxide-soaked chips to remove toxic materials. The use of coal tar itself for

Figure 2-1: A map of Site 24 showing basic site features (as of January 1989) and general direction of groundwater flow.



its organic constituents grew to major importance in the chemical manufacturing industry during the World War II era (Anastos et al., 1986).

When the commercial value of by-products of the manufactured gas industry were not utilized, these chemicals were commonly disposed of on the site of the town gas plant. Since there are well over 1,000 gas plants across the country, coal tar contamination of groundwater and soil has become a frequent and major concern.

Coal tar is a black tarry substance, containing an estimated 10,000 compounds (Enzminger and Albert, 1987). These compounds range from single ring aromatics such as benzene, toluene and xylene and two-ring compounds such as naphthalene to aromatic compounds with more than 20 rings. A table showing the major constituents of two coal tars is shown in Table 2-I. Sixteen polynuclear aromatic hydrocarbons (PAHs) found in coal tar are on the U.S. Environmental Protection Agency (EPA) priority pollutant list. Table 2-II lists the constituents of coal tar found in evidence at Site 24.

For reasons of simplicity, only one constituent of the coal tar was chosen to be tracked during our study. Naphthalene was selected as the compound of interest because it is highly soluble (compared to other coal tar constituents), conservative (low octanol-water partition coefficient (K_{ow})), not highly volatile (considered only semi-volatile), and abundant in the site. Pertinent physical data on naphthalene are shown in Table 2-III. Though not as toxic as some other polyaromatic hydrocarbons, naphthalene is on the U.S. EPA priority pollutant list. Naphthalene is also is the key ingredient of mothballs, giving mothballs their distinctive odor.

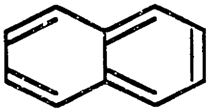
Table 2-I: Constituents of Two Coal Tars (Enzminger and Albert, 1987)

Constituent	Coke-Oven Tar (%)	Lurgi-Gasifier Tar (%)
Acenaphthene	1.05	0.57
Anthracene	0.75	0.32
Benzene	0.12	0.02
Carbazole	0.60	0.22
o-Cresol	0.25	1.14
m-Cresol	0.45	1.83
p-Cresol	0.27	1.51
Diphenylene oxide	----	0.57
Ethylbenzene	0.02	0.04
Fluorene	0.64	0.62
High-boiling tar acids	0.83	11.95
Medium-soft pitch	63.5	33.1
α-Methylnaphthalene	0.65	0.63
β-Methylnaphthalene	1.23	1.05
Naphtha	0.97	3.02
Naphthalene	8.80	2.01
Phenanthrene	2.66	0.28
Phenol	0.61	0.97
Styrene	0.02	0.01
Tar bases	2.08	2.50
Toluene	0.25	0.05
o-Xylene	0.04	0.05
m-Xylene	0.07	0.07
p-xylene	0.03	0.03
Xylenols	0.36	5.55

Table 2-II: Coal Tar Constituents Found in the Groundwater at Site 24

Semivolatiles	Volatiles
naphthalene	toluene
2-methylnaphthalene	ethyl benzene
1-methylnaphthalene	benzene
acenapththalene	xylenes
acenaphthene	styrene
dibenzofluoranthene	indene
fluoranthene	trimethyl benzene
phenanthrene	indan
anthrene	
fluorene	
pyrene	
benzo(a)anthracene	
chrysene	
benzo(b)fluoranthene	
benzo(k)fluoranthene	
benzo(a)pyrene	
indenopyrene	

Table 2-III: Physical Properties of Naphthalene



The chemical structure of naphthalene.

Molecular Weight	128	Enzminger et al., 1987
Log K_{ow}	3.00- 4.00	Enzminger et al., 1987
Liquid Solubility (at 10°C)	87.9 ppm	Groher, 1990
Melting Point	80°C	Groher, 1990
Henry's Law Constant	1.15×10^{-3} atm-m^3/mol	Lyman et al., 1982
Specific Gravity	1.152	Verschueren, 1977

2.3 Characteristics of Site 24

2.3.1 Soil Quality

One quality that makes Site 24 a good research site is that the subsurface is for the most part homogeneous and sandy. This is a fairly typical northeast aquifer and has allowed for easy hand augering of piezometers and almost perfect drilling conditions. There are no stones or structures in this site that could impede drilling. Several tests of soil properties were run by Atlantic Environmental Services (AES). It is to be expected, and will later be shown, that the contamination is mainly found in the most conductive soil layers, the fine to medium size sand.

2.3.2 Hydraulic Conductivities, Soil Types and Porosity

In-lab permeability tests were conducted by Woodward-Clyde Consultants for AES on five soil core samples taken from MW 11 at Station 8. From these data, hydraulic conductivities (K) were calculated for each sample. These results give values of K in cm/sec that range over five orders of magnitude. A table summarizing the permeability test data is provided in Table 2-IV. These same core samples were also analyzed for particle size distribution in order to categorize their soil types. The results of these tests are shown in Figure 2-2. The porosity of the fine to medium size sand, as determined by Core Laboratories for AES, was 34.7 percent or 0.347.

Table 2-IV: A Summary of Permeability Tests

Sample No. CC-5	Orientation	M.C.$_o$ %	γ d pcf	S$_o$ %	M.C.$_f$ %	S$_f$ %	K cm/sec
STA-8 20-21	Vertical	34.6	81.8	88.0	32.4	99.8	2.68×10^{-7}
STA-8 20-21	Horizontal	44.3	74.9	95.9	40.2	99.7	5.96×10^{-6}
STA-8 20-21	Horizontal	53.0	72.3	100.0	45.8	98.9	3.37×10^{-6}
STA-8 10-11	Vertical	15.2	96.3	56.4	23.5	99.9	1.06×10^{-2}
STA-8 16-17	Vertical	21.5	98.4	83.9	22.7	99.9	1.78×10^{-3}

Note: Specific gravity was assumed to about 2.65 to 2.70 for computation of degree of saturation.

Where:

M.C. = Initial or final water content

γ d = Initial dry density

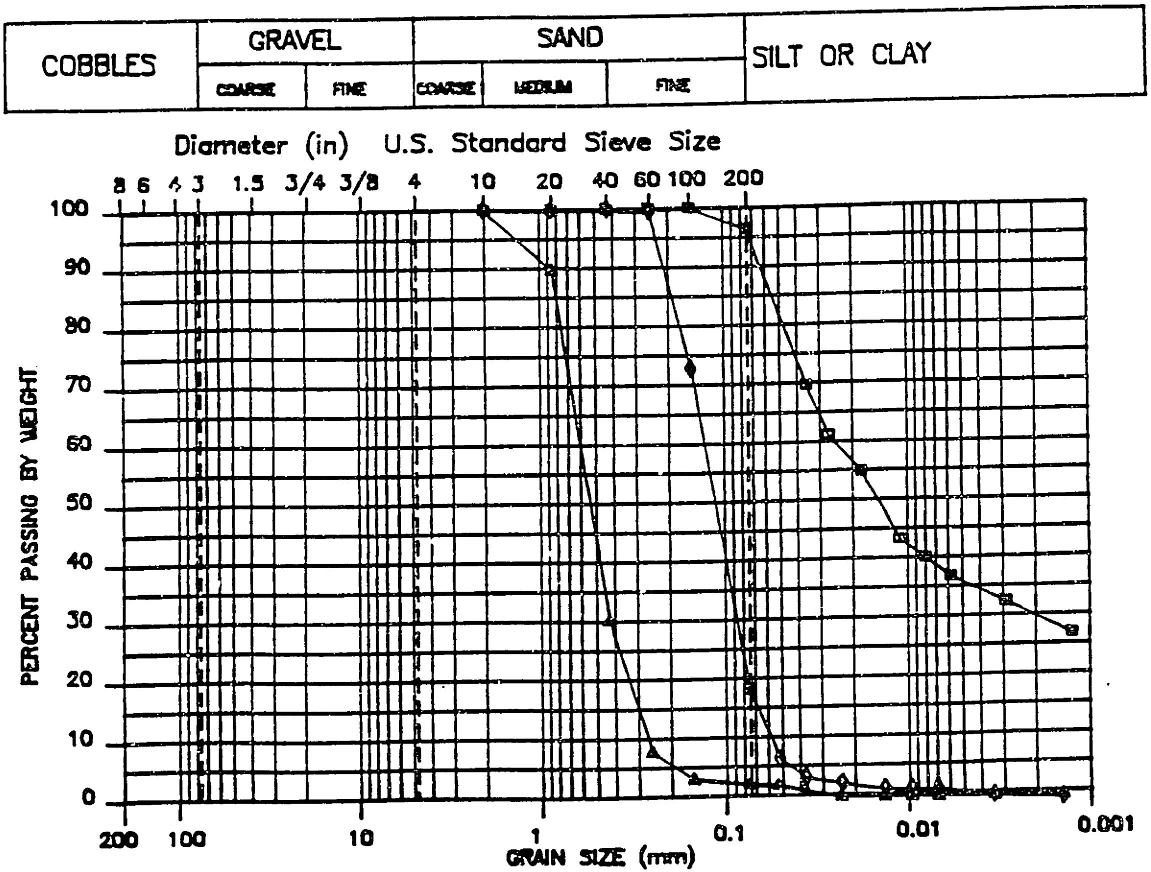
S = Initial or final degree of saturation

K = Coefficient of permeability corrected to 20°C

Field pump tests to determine saturated hydraulic conductivity were performed by AES in mid-1988 at four monitoring wells. Values of hydraulic conductivity calculated from these tests are shown in Table 2-V.

Slug tests were performed at Site 24 by AES in mid-1989 at 17 different monitoring wells. The locations of these well stations are shown in Figure 3-9. The hydraulic conductivities were calculated from the normalized drawdown versus time curves and these results are shown in Table 2-VI. The Horslev method for interpreting slug test data from wells that are only screened over a short distance was used to make these calculations (Freeze and Cherry, 1979). In this table and the previous table depths are in units of feet below ground and hydraulic conductivity is in units of cm/sec. A reasonable value of K for the medium to fine sand in which the naphthalene is primarily found in Site 24, is 7.24×10^{-3} cm/sec. This number is an average of the data marked with an asterisk in Tables 2-V and 2-VI from all the field tests done on sands in the range medium to fine.

Figure 2-2: Particle-size distributions for three soil samples.



JOB NUMBER : 88C2223

JOB NAME : ATLANTIC ENVIRONMENTAL SERVICES

SYM	BORING#	SAMPLE#	DEPTH	DESCRIPTION	W(%)	W_L(%)	W_P(%)
□	CC-5	STA 8	20-21	GRAY CLAYEY SILT(ML)			
△	CC-5	STA 8	10-11	TAN MEDIUM TO FINE SAND.			
◇	CC-5	STA 8	16-17	BROWN GRAY SILTY FINE SAND.			

Table 2-V: A Summary of Results of Pump Tests Performed in 1988

Sta.	Well	Depth	Formation where screen is located	K
2	10	6-11 fbg	medium to coarse sand*	3.51×10^{-4} cm/sec
5	13	15-17	fine sand and silt with clay	4.12×10^{-5}
6	20	17-22	fine sand and silt*	1.16×10^{-4}
10	19	17-22	very fine sand*	2.63×10^{-4}

(* Indicates value used in average.)

Table 2-VI: A Summary of Results of Pump Tests Performed in 1989

Sta.	Well	Depth	Formation where screen is located	K
3	6	12.5-17.5 fbg	coarse sand	1.28×10^{-3} cm/sec
3	7	7-12	coarse sand	1.14×10^{-2}
5	13	15-17	medium sand*	4.41×10^{-3}
5	15	6.5-9.5	coarse sand	1.37×10^{-2}
5	25	12.5-15	medium sand*	2.16×10^{-2}
7	8	16-21	fine sand*	5.08×10^{-5}
7	9	11-16	medium sand*	3.88×10^{-3}
9	16	14-16	medium sand*	7.89×10^{-3}
9	17	9-12	medium sand*	4.18×10^{-2}
9	12	16-21	fine to medium sand with clay*	9.96×10^{-4}
10	14	8-10	coarse sand	4.41×10^{-2}
13	27	32.5-3	silt with clay	8.83×10^{-4}
13	28	27.5-30	fine sand with clay*	2.95×10^{-3}
15	31	23.5-26	silt with fine sand	1.24×10^{-3}
15	32	20.5-23	fine sand*	4.97×10^{-3}
19	36	20.5-23	silt with clay	8.21×10^{-4}
19	37	17.5-20	fine sand*	4.78×10^{-3}

(* Indicates value used in average.)

2.4 Groundwater Flow Field

2.4.1 Horizontal Head Gradients

A simplified way of examining the hydraulic head field in an aquifer is to assume hydrostatic conditions. Under this assumption there are no vertical head gradients, reducing the head field to simply the elevation of the water table. Only data from piezometers and monitoring wells whose screens straddle the water table are considered in such an approach.

Many piezometers were installed by AES and MIT over the period of study at Site 24. Water table levels were taken throughout the site at six different times during the study in order to develop a more complete picture of the water table as more piezometers were installed. The most thorough set of head data was taken on May 12, 1989. A map of the site showing the locations of the piezometers installed by this date, and a plot of the water level contours drawn from head data taken from those locations are shown in Figures 2-3 and 2-4. These two figures were drawn up by Atlantic Environmental.

An average range of the horizontal head gradient, estimated from the map in Figure 2-4, is about 1 foot drop in head per 50-120 feet of distance, or 0.02 - 0.008. This range represents the natural head gradient over most of the area between the source and the seeps. The gradient is much steeper at the seeps, which serve as a groundwater discharge zone.

2.4.2 Vertical Head Gradients

It may be easier to look at the head field as if it were two-dimensional, but it is important to consider gradients of hydraulic head in the vertical. Significant vertical head gradients can be responsible for sinking or raising of contamination in the aquifer.

Vertical head gradients are significant at Site 24. Table 2-VII shows the vertical head gradients at 10 well stations (well clusters), calculated from the distance between the centers of two monitoring well screens, and their water level difference (water levels taken

Figure 2-3: A map of Site 24 showing the locations of all piezometers where water level measurements were taken on 5/12/89 (from AES).

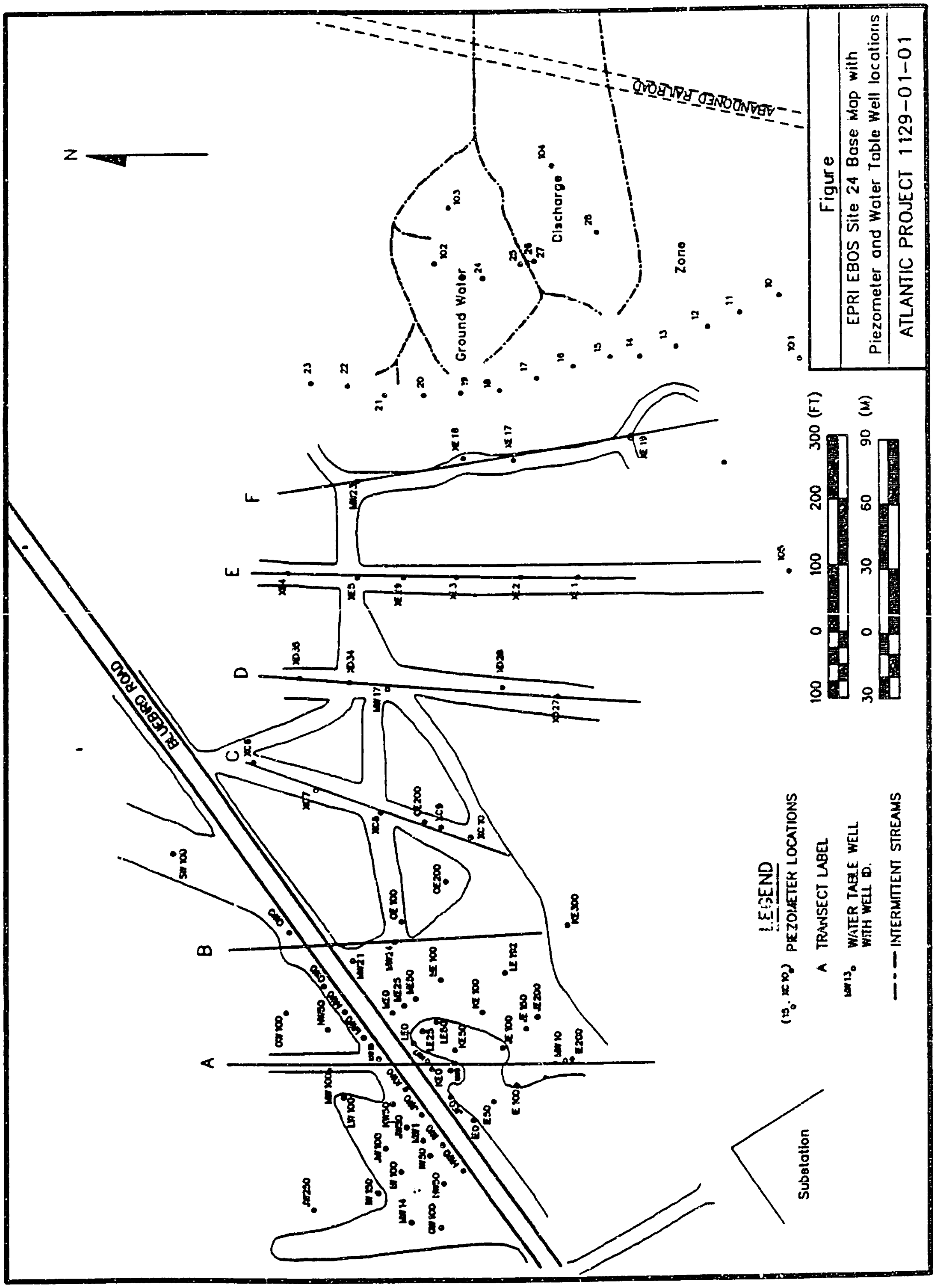
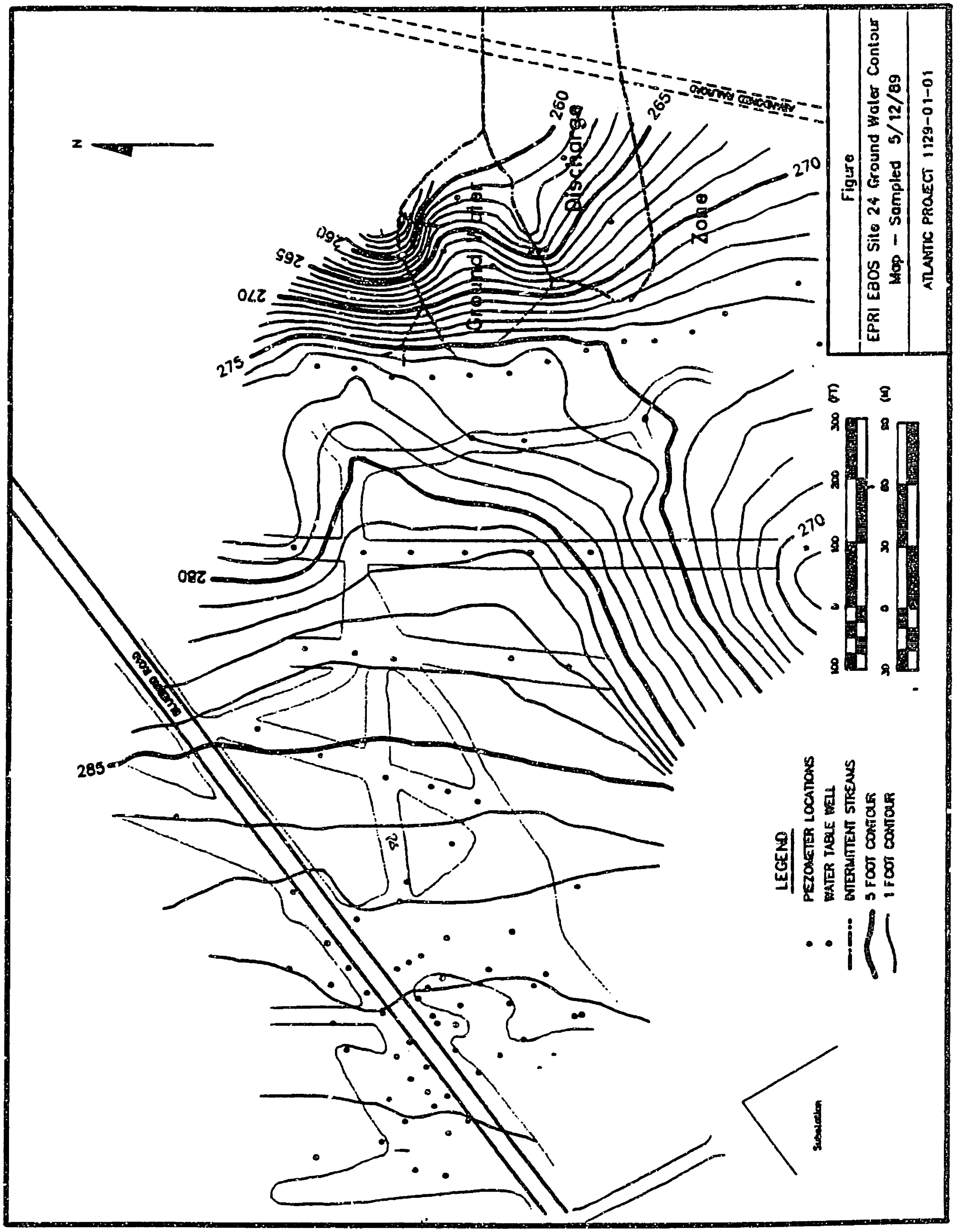


Figure 2-4: Water level contours and groundwater flow lines derived from piezometer data taken on 5/12/89, elevations shown in feet (from AES).



May 12, 1989, all distances in feet). Except for a few extreme cases, at most well stations the vertical head gradient is 0.04 or lower, up to twice the horizontal head gradient. Vertical gradients are particularly large at Stations 8 and 11. Because the error in taking water level measurements can be up to one tenth of an inch, these figures are not very reliable, but provide a qualitative indication of the potential importance of vertical flow effects.

The elevation of the water table changes during the year with the seasons. Figure 2-5 displays such changes as they have occurred at Site 24. Water table elevation changes, as high as three feet between winter and spring, could possibly have an effect on the shape of the plume. These changes are brought about by such environmental influences such as rain, snowmelt and evaporation.

2.5 Groundwater Velocity and Travel Times

It is convenient, and probably not unreasonable, to assume that after 28 years, the plume at site 24 has reached steady-state. This means that the contamination has spread out in three dimensions as much as it is going to and will stay that way, given that the source, flow field and precipitation patterns do not change. The presence of the seeps, which act as continuous sinks, greatly influence the direction of groundwater flow. The shape of the plume may vary only slightly in the short-term due to temporary variations in the flow field mentioned in the previous section.

Using the Darcy Equation given below an approximate value of the groundwater velocity and travel times can be calculated:

$$\nu = K/n \; dh/dl$$

Estimates for K (hydraulic conductivity), n (porosity) and dh/dl (head gradient) that have already been described are shown below with the resulting groundwater velocity (ν).

Table 2-VII: Calculations Showing Vertical Head Gradients at 10 Well Stations at Site 24

Sta.	Well	h	Depth	Between Wells	dh	dl	dh/dl
1	1	288.92	8.5	1-2	0.13	4	0.0325
	2	288.79	12.5	2-3	0.01	4	0.0025
	3	288.78	16.5	1-3	0.14	8	0.0175
3	6	288.36	15	6-7	0.05	5.5	0.0091
	7	288.41	9.5				
7	8	286.69	18.5	8-9	0.26	5	0.0520
	9	286.95	13.5	9-24	0.03	5	0.0060
	24	286.98	8.5	8-24	0.29	10	0.0290
8	11	283.77	28.5	11-21	3.15	17	0.1853
	21	286.98	11.5	21-22	2.6	10.5	0.2476
	22	284.32	22	11-22	0.55	6.5	0.0846
9	12	283.35	18.5	12-16	0.1	3.5	0.0286
	16	283.45	15	16-17	0.17	4.5	0.0378
	17	283.62	10.5	12-17	0.27	8	0.0337
10	14	289.41	9	14-19	-0.15	10.5	-0.0143
	19	289.56	19.5				
11	23	280.00	14.5	23-26	4.46	15	0.2973
	26	275.54	29.5				
13	27	271.75	33.75	27-28	0.21	5	0.0420
	28	271.96	28.75				
15	31	279.60	24.75	31-32	0.09	3	0.0300
	32	279.69	21.75				
19	36	285.54	21.75	36-37	0.12	3	0.0400
	37	285.66	18.75				

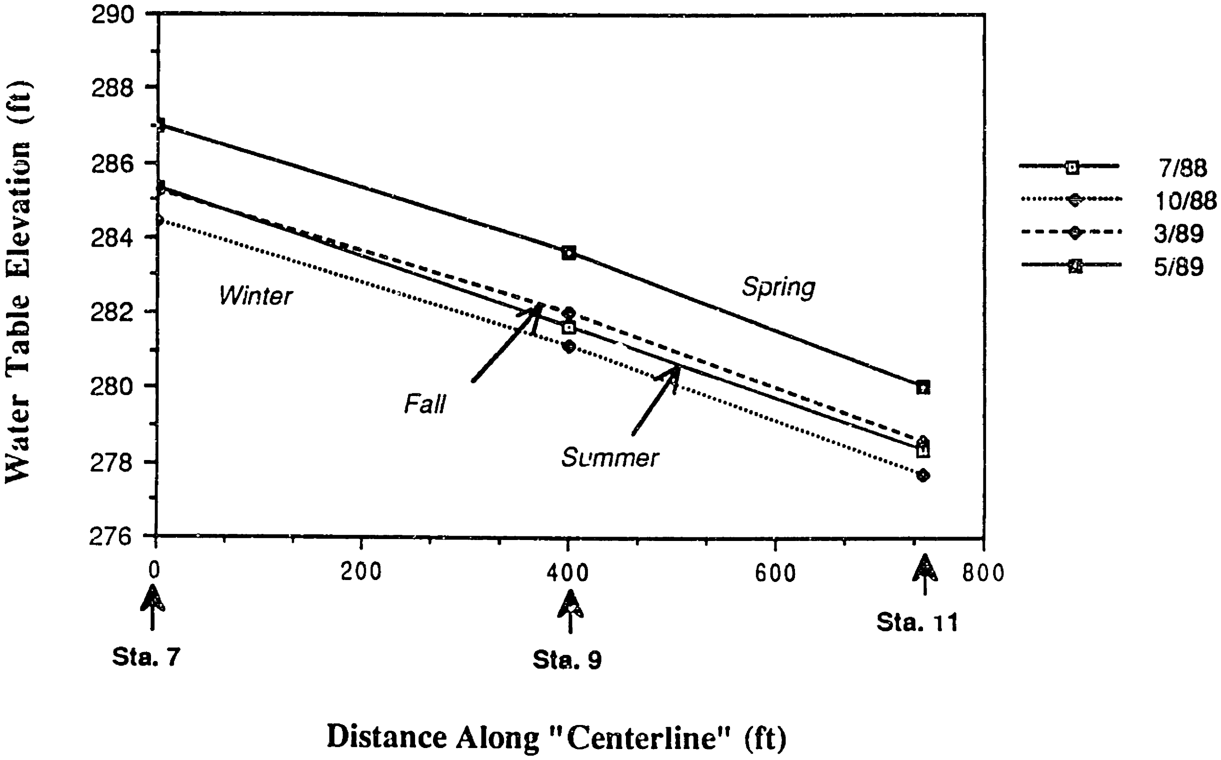


Figure 2-5: Variation of water table elevation with time at Site 24.

Based on these values it should take 2.7 to 6.8 years for contaminants to be transported from the source to the seeps (1150 ft = 350 m). This verifies the assumption that the plume is in steady-state.

$K = 7.24 \times 10^{-3}$ cm/sec

$n = 0.347$

$dh/dl = 0.02$ to 0.008

$\nu = 3.6$ to 1.4 cm/day

2.6 Chronology of Activities at Site 24

In total, fourteen multi-level samplers have been constructed and installed at Site 24. The two PVC-tubed MLSs are called P-1 and P-2, and the twelve aluminum (metal) samplers are called M-1 through M-12, numbered in order of installation. Table 2-VIII below outlines the chronology of sampling and installation events carried out at Site 24.

Table 2-VIII: Chronology of Installation and Sampling Events in the Site 24 Program

Date	Activity
April 10, 1989	Installation Round 1: P-1, P-2, M-1, M-2
June 6, 1989	Sampling Round 1: P-1, P-2, M-1, M-2
Sept. 5, 1989	Sampling Round 2: P-1, P-2, M-1, M-2
Sept. 11, 1989	Installation Round 2: M-3 thru M-8
Oct. 5, 1989	Sampling Round 3: M-1 thru M-8
April 3, 1990	Installation Round 3: M-9 thru M-12
April 26, 1990	Sampling Round 4: M-9 thru M-12

Chapter 3

Multi-level Sampling

3.1 Previous Large-scale Applications of MLSs

Multi-level samplers are known for their use at three major research sites, at the Borden Air Force Base in Ontario, Canada (Cherry et al., 1983), at the Otis Air Force Base on Cape Cod, Massachusetts where over 1000 MLSs were installed (Garabedian, 1987), and at the Columbus Air Force Base in Mississippi (TVA, 1988). At all the sites, large-scale, natural-gradient tracer tests were conducted to examine the transport of reactive and nonreactive tracers. The Borden experiment was conducted primarily by the University of Waterloo in Ontario. The Cape Cod experiment was conducted by the U.S. Geological Survey (U.S.G.S.) of Marlboro, Massachusetts. The Columbus project, also known as the Macrodispersion Experiment (MADE) was executed by the Tennessee Valley Authority and funded by EPRI. The aquifer at the Columbus site is quite heterogeneous, with terraced deposits occurring in lenses. Like the Borden and Cape sites, Site 24 is an undeveloped site with a sandy aquifer.

Multi-level samplers were used exclusively in these tracer tests to collect tracer concentration samples because sharp vertical concentration gradients were anticipated (monitoring wells were used for aquifer pumping tests and to inject the tracers). At the Columbus site, vertical hydraulic gradients are locally as high as seven percent. Unlike monitoring wells, multi-level samplers, when applied correctly, can capture the vertical variations in solute distributions. The application of multi-level samplers at one site does not guarantee their success at another site, however. At every new site, the MLS design must meet specific criteria, and in situ tests must be run to verify that the samplers have been installed correctly. Criteria to be considered when designing a multi-level sampler, and verification tests that can be run will be described later in this thesis.

Unique to my project is the MLS design developed for use at Site 24. In the three major projects involving MLSs previously mentioned, MLSs were sampled for inorganic tracers. The challenge at Site 24 was to build MLSs that could be sampled to determine accurate concentrations of organic contaminants in the groundwater. Several commercially available MLSs will be described in the section, "MLS Designs Developed for Industry and Research".

3.2 Data Collection from a MLS Versus a Monitoring Well

The theory behind the advantage of multi-level sampling over sampling from conventional monitoring wells is that MLSs take samples from small, discrete zones in the aquifer while samples from monitoring wells are an average of the concentrations over the length of the screen. The picture shown in Figure 3-1 demonstrates this difference. In this figure, the monitoring well whose screen only spans part of the plume will give data less than the average concentration of the part of plume it intersects because uncontaminated water from above the plume also enters the well. With the effort it takes to drill one hole, more information may be obtained by installing a MLS than a monitoring well (in some situations more data may not be necessary or economically expedient or MLS use may not be appropriate- these cases will be described later).

Monitoring well data can only be used to estimate average, not absolute concentrations of a contaminant in an aquifer. For example, if the concentration goes from 0 to 20 over the length of the screen or is at a constant value of 10 over the length of the screen, the results will be the same, an average of 10.

When the ports of a multi-level sampler are pumped simultaneously during sampling (in a homogeneous, isotropic, infinite, unconsolidated aquifer), the sampling area should be from halfway between the sampled port and the ports above and below it. The diagram in Figure 3-2 shows the approximate flow patterns to a MLS during sampling. Ideally, the ports of a MLS will only sample from their particular horizontal zone.

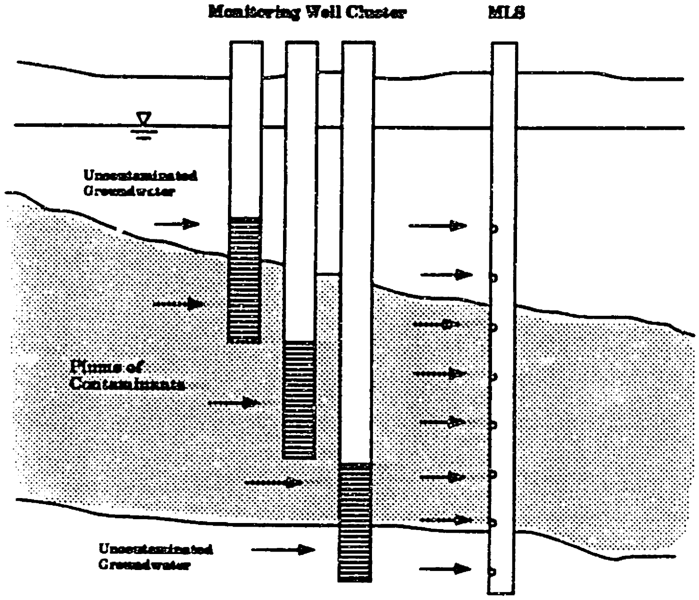


Figure 3-1: A multi-level sampler and a cluster of monitoring wells intersecting a contamination plume.

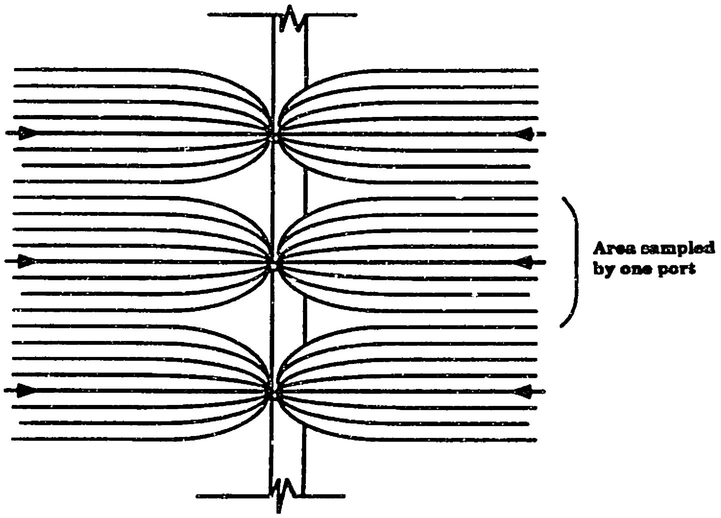


Figure 3-2: Ideal groundwater flow lines to a MLS induced by pumping during sampling.

3.3 MLS Designs

3.3.1 MLS Designs Developed for Industry and Research

Because of the success of the MLS design used for the large-scale tracer test at the Borden site, the design was implemented at the Cape site. The MLS used at these sites is one of several basic styles of multi-level sampler, called the cased MLS. In the cased MLS, an outer casing (usually PVC) has small port holes drilled into it. Thin collection tubing (may be metal or plastic) extends from each sampling port hole, through the protective casing pipe, up to the ground surface. Screening materials are used to cover each sampling port to prevent soil from entering and clogging the tubing. The casing is capped at the bottom to prevent the contaminated aquifer water from entering the MLS and possibly leaking into the collection tubing.

The cased MLS design was first reported by Pickens et al. in Ground Water in 1978. Figure 3-3 shows the design and materials proposed by Pickens et al. and used at the Borden site. The MLS used at the Cape site only differed in sampling port configuration. In the Cape MLSs, the plastic tubing (in both cases polyethylene) extended outside the pipe about two inches from each port hole. These ends were wrapped in nylon cloth and bound with stainless steel wire to the pipe. Teflon tubing was used in the cased samplers at the Columbus site and also used nylon filters. Experimental MLSs tested at the Columbus site were constructed with stainless steel casings so that the samplers could be driven into the ground.

Although the cased MLS is a very popular style due to the ease of its construction, it is limited in that a peristaltic pump must be used to draw the groundwater up from the thin tubing. The depth to which groundwater can be sampled from a cased MLS is thus limited to 25 feet below the ground surface (see subsection "Depth of the Water Table Below Ground" in the section "General MLS Design Criteria").

The second basic style of MLS design is called a bundle piezometer. In this design the collection tubes are attached to the outside of the casing and the casing only serves to give the rigidity necessary to maneuver the sampler for installation. A bundle piezometer was used in the Borden aquifer study, a schematic of which is shown in Figure 3-4. Depending on the width of the tubing used, a submersible pump, a bailer and a peristaltic pump may be used to sample a bundle piezometer.

A third and less common technique of multi-level sampling is done by collecting samples from discrete zones along the screen of a monitoring well. Zones along the screen are kept separate by way of flexible or inflatable packers or seals. Ronen et al. present a multi-level sampling unit that after being dropped down a monitoring well, may sample from as many as 38 ports set at 3 cm intervals (patents pending). The sampler consists of a PVC casing set with dialysis cells. After this MLS is kept in a monitoring well over a period of time, chemical concentrations are established in each cell in equilibrium with the concentrations of the chemicals in the groundwater at those locations. Equilibration periods can vary from 1 day to 1 week depending on the diffusion coefficient of the compound in question (Ronen et al., 1987). A diagram of the MLS described by Ronen et al. is shown in Figure 3-5.

A multi-level sampler designed for groundwater monitoring in fractured rock is commercially available from Solinst Canada Ltd., Williams, Ontario, Canada. Called the Waterloo System, the Solinst MLS is modular, and allows up to 6 sampling points in a 3 inch borehole, and up to 10 sampling points in a 4 inch borehole. It is specifically designed for use in a borehole in a solid formation because of the unique packers attached to the MLS that isolate each sampling point within the borehole. These packers are made of a solid but flexible sealant that expands on contact with water. A detailed description of the MLS design marketed by Solinst can be found in an article by Cherry et al. in Groundwater Monitoring Review (Cherry and Johnson, 1982).

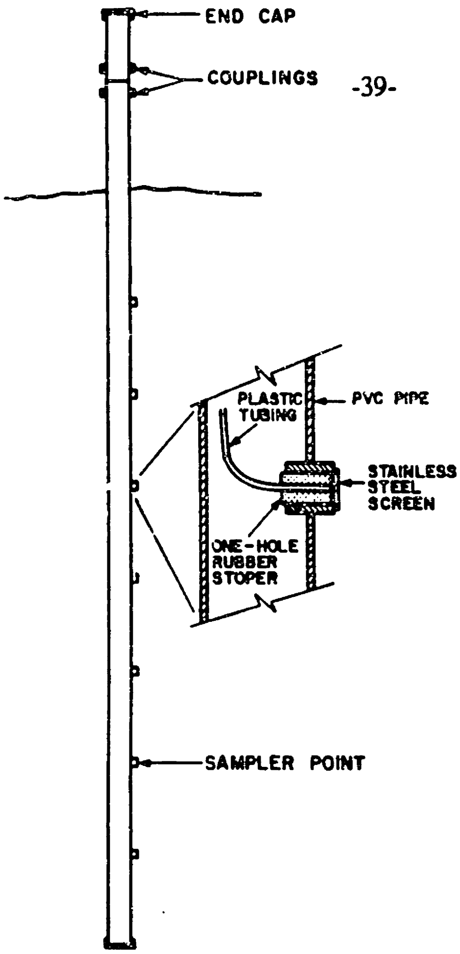


Figure 3-3: Schematic diagram of MLS Proposed by Pickens et al. and used at the Borden site (Cherry et al., 1983).

Barcad Systems, Inc. of Concord, MA has also developed a system for sampling from multiple levels within the same borehole. In the Barcad system, thin sampling wells are installed one at a time to specific levels within a borehole (lowest to highest). Bentonite seals are placed between each sampler within the borehole to isolate each one, and they are commonly packed with a sand. At the bottom of each mini-well is a 16" long sampler, 1.5" in diameter, with a porous filter to let in only water. Groundwater is drawn up to the ground from the sampler via a tube inside a tube by an unusual technique called gas drive. By a pattern of forcing and venting pressurized gas in the outer tube, and with the assistance of a check valve located in the top of the sampler body, groundwater may be brought up the inner tube. The advantage of this positive pressure system is that samplers may be installed to any depth. Barcad samplers may be made out of a number of different materials, depending on the contaminants to be sampled for.

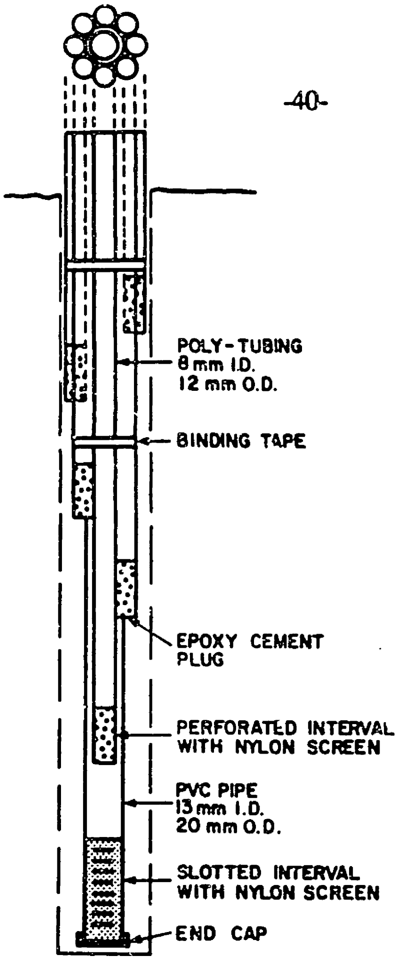


Figure 3-4: Schematic diagram of the bundle piezometer used at the Borden site (Cherry et al., 1983).

3.3.2 The MLS Design Developed for Use at Site 24

The MLS design implemented at Site 24 is the cased MLS, similar to the MLS from the U.S.G.S. Cape Cod study. A typical PVC casing was used, and two different materials were used for collection tubes, PVC and aluminum. PVC was used for collection tubes in two of the initial MLSs because it is inexpensive. Its use was discontinued, however, because it was thought that adsorption was artificially reducing the measured concentration values. In addition to collecting more representative groundwater samples, the aluminum MLSs were easier to assemble. A schematic of the sampling port configuration I developed for use with the rigid aluminum tubing is shown in Figure 3-6. In total, fourteen multi-level samplers have been constructed and installed at Site 24 to date, two PVC-tubed MLSs and twelve aluminum samplers.

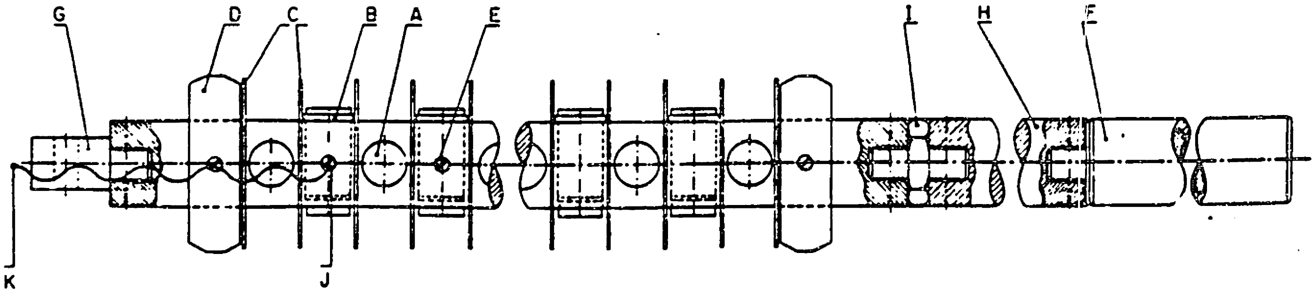


Figure 3-5: Schematic diagram of the reusable MLS proposed by Ronen et al. to be placed inside a monitoring well (Ronen et al., 1987).

(A) crisscrossed holes in PVC (B) dialysis cells (C) flexible rubber seals (D) PVC guiding rings (E) nylon screws (F) PVC-coated weight (G) upper holding segment (H) additional rod segments (I) double-screw connectors (J) stainless steel screw (K) insulated stainless steel wire.

The brass elbows are Swagelok© fittings, and the nuts are caps from these fittings, machined into nuts. The threaded ends of the elbows that are held to the PVC by the nuts are sanded down to prevent the edges from damaging the screening materials. The resulting thin nut and sanded end make a sampling port that has no sharp edges and only extends beyond the PVC by about 3/16 inch. The PVC is threaded, schedule 40, purchased in ten foot sections with screw-on caps for the bottom section of each MLS. The aluminum tubing is 60/61 grade, 3/16 inch diameter with an .035 inch wall. The aluminum tubes came in twelve foot sections and are connected with brass Swagelok© unions. The hose clamps used to fix the screening materials over the sampling ports are size 36, all stainless and are the largest protrusions on the sampler at 7/16 inch. The first four MLSs were designed with 8 sampling ports at 2 foot intervals, spanning a depth of 14 feet. Because a couple ports of the first MLSs inadvertently ended up above the water table, the next six MLSs were made with 10 sampling ports at 1 foot intervals, spanning a depth of 9 feet. Practical difficulties were encountered sampling 10 ports, since three pumps (of a maximum of 4 pump heads each) had to be run simultaneously, so the final four MLSs were built with 8 ports (only two 4-head pumps needed).

3.4 General MLS Design Criterion

A concise summary of the criterion considered in the design of multi-level samplers for Site 24 is presented in Table 3-I. In addition to the obvious cost considerations, chemical and physical properties of the soil and groundwater must be carefully looked at to arrive at the optimal MLS design. Most of the considerations described below were arrived at in retrospect due to the accelerated pace of this project.

3.4.1 Soil Collapsibility

Of primary concern in the use of the MLS is soil collapsibility. Based on the experiences at Borden, on the Cape, and looking back on this project, where the soil is unable to collapse around the MLS (i.e. in clay), samples taken will not be representative of the aquifer. Unconsolidated soils are required for accurate implementation of MLSs. Because we questioned the ability of the soil at Site 24 to fully collapse around a bundle piezometer-style MLS, a cased MLS was used. Small-scale tracer tests can be performed in the field to check for gaps around the sampler.

3.4.2 Contaminant (Materials) Considerations

Materials considerations are directly influenced by the types of contaminants to be looked for. Our design specifications had to meet specific criterion that the other projects did not have; our multi-level samplers had to take reliable samples of the organic compound naphthalene. Our MLS had to be built in such a way as to prevent leaching of organics from

Figure 3-6: Schematic detail of the sampling port area of the cased MLS used in this project.

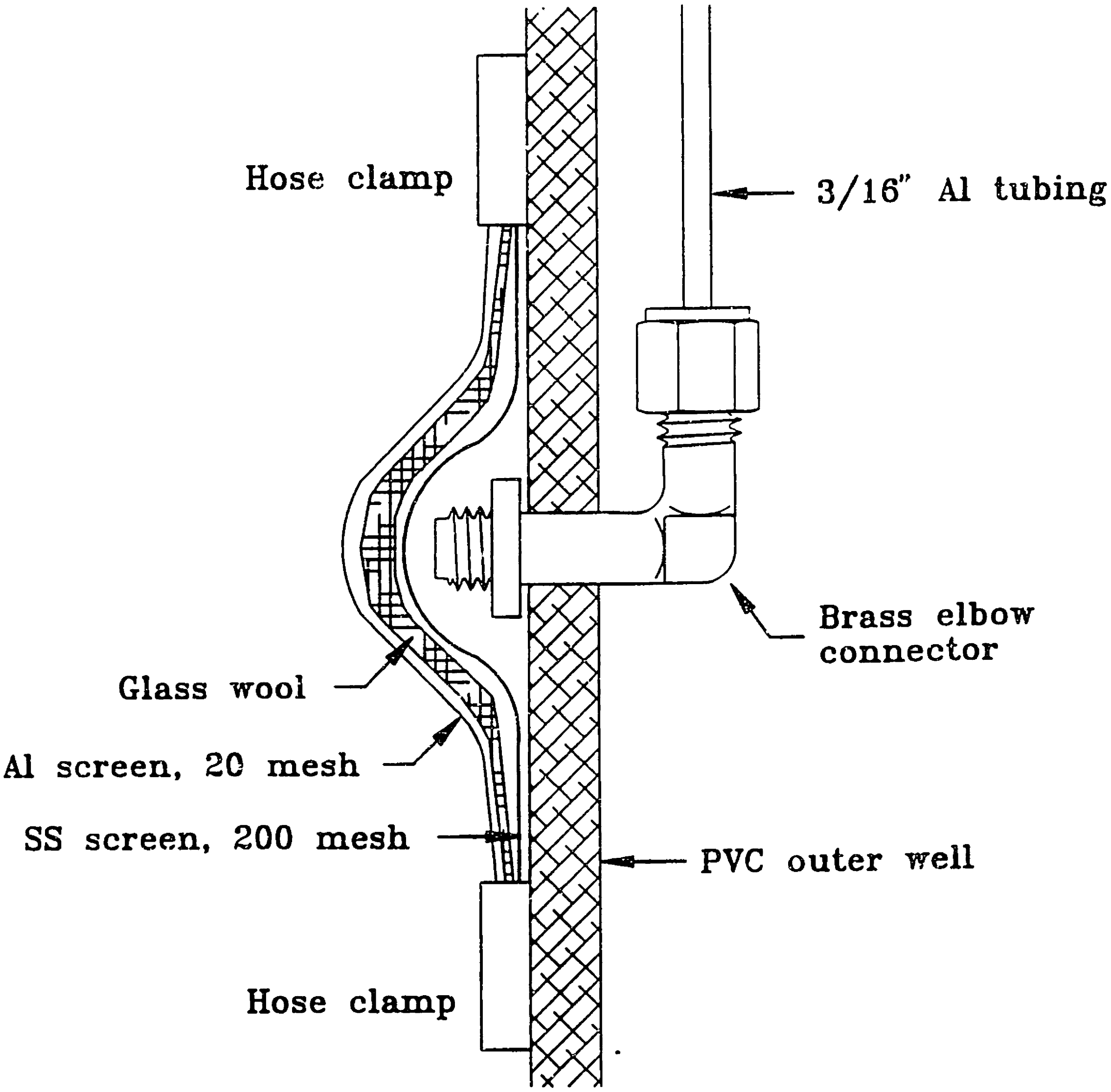


Table 3-I: Criterion Considered in the Design of MLSs for the Site 24 Study

1. Collapsibility of soil
2. Types of contaminants to be looked for
3. Soil particle size
4. Costs
5. Installation techniques available
6. Depth below ground surface of groundwater to be sampled
7. Ease of construction
8. Final weight of sampler

the sampler materials into the sample water, and prevent naphthalene from adsorbing out of the sample water and onto the materials. No glue, tape or caulk could be used, and the amount of plastic touching the sample water had to be kept to a minimum. For this reason, aluminum, brass, stainless steel and glass were used in our samplers. The sections of PVC casing we used were joined by flush threads. Similarly, if metals are to be sampled for, metal should not be used in the sampler because of the threat of leaching.

The literature is undecisive about the materials that should be used when sampling for organics. Pettyjohn et al. claim that "metals may strongly adsorb organic compounds" and use as an example that "DDT...is strongly adsorbed even by stainless steel" (Pettyjohn et al., 1983). They admit though that experimental data simply does not yet exist on the potential reactions between the wide range of materials and organic compounds that are available. One of the most prolific researchers in the field of appropriate sampling materials is M. J. Barcelona. One of his more recent studies (Holm et al., 1988) confirmed that gases could diffuse in significant quantities through lengths of polymeric tubing, potentially increasing dissolved oxygen, decreasing dissolved CO_2, and altering pH. In another study by Barcelona (Barcelona and Helfrich, 1986), differences in sampling of organics from monitoring wells made from Teflon, stainless steel and PVC were not predictably high or low for any of the materials. But at another site in the study, organic concentrations from stainless steel were found to be higher than Teflon, which was in turn higher than PVC. Two lists of (rigid and flexible) materials preferred for use in groundwater sampling devices based on their inertness, are shown in Table 3-II (Nielsen and Yeates, 1985). These lists were derived from a compilation of eight different materials studies, including Pettyjohn's and three done by Barcelona (Barcelona et al. 1984; Barcelona et al. 1983; Barcelona 1983; Curran and Thompson 1983; Miller 1982; Sosebee et al. 1982; Pettyjohn et al. 1981; Scalf et al. 1981).

Table 3-II: Preferred Materials for Use in Groundwater Sampling Devices (Nielsen and Yeates, 1985).

(A) Rigid Materials	(B) Flexible Materials
1. Teflon	1. Teflon
2. Stainless steel 316	2. Polypropylene
	3. Flexible PVC/Linear
3. Stainless steel 304	Polyethylene (PET)
4. Polyvinylchloride (PVC)	4. Viton
5. Low-carbon steel	5. Conventional PET
6. Galvanized Steel	6. Tygon
7. Carbon steel	7. Silicone/Neoprene

The same materials considerations that apply to MLS construction apply to the apparatus required for extracting groundwater from the MLSs. The tubing, devices, and pumps used for sampling must not alter the sample chemistry. A description of the sampling apparatus used at Site 24 will be presented in a later section.

3.4.3 Minimum Soil Particle Size and Colloids in the Groundwater

Preservation of colloids (size range nanometers to microns) is an important consideration when sampling groundwater because these suspended particles can carry a significant fraction of the contaminant. Colloids of concern in aquifers are usually made of metal oxides and so they are usually present when the level of dissolved oxygen and mineral content in the groundwater is high. In aquifers where colloids are assumed to be present, care should be taken not to put too much filter material on the samplers. At the same time, screening materials should be fine enough to prevent most of the small particles from entering the sampling ports to prevent clogging. In general it is desirable not to filter groundwater samples in order to get the most representative sample of the mobile solute.

It was determined by Professor Phil Gschwend, a colloids specialist at the MIT Parsons Lab, that colloids are not present in the Site 24 aquifer. If they had been, the glass wool on our samplers might have been omitted as a screening material.

3.4.4 Costs

The most expensive aspect of multi-level sampling is the number of samples that need to be analyzed. In order to get the most out of each MLS, the number, spacing and elevation of the sampling ports must be carefully determined prior to construction. The first step is to decide where on the site the MLS is going to be installed. Next, the water table elevation, and the top and bottom of the plume at this point must be estimated. The ports are then situated along the MLS to intercept as much of the plume as possible, with one or two ports placed just outside the approximate edges of the plume, in order to try to resolve these edges. All ports are located below the water table, so that materials and effort are not wasted on installing sampling ports which will not be able to draw water.

At Site 24, it was difficult to place ports "below" the naphthalene plume because the bottom of the plume went right up against the clay layer. The Round 2 MLSs were constructed taking the head data into account, and the spacing of ports on Round 3 MLSs were based on head, and vertical concentrations from previous soils, monitoring well and MLS data. While this technique of setting the dimensions of the MLS for its particular location on the site takes much more planning and work than simply building samplers and then deciding where to put them, it produces more functional sampling ports and fewer repetitive non-detects. Glancing at the MLS data in Appendix A, it is evident that the earlier MLSs built had more multiple non-detects and dry ports than the later MLS designed with more data available.

An outline of the costs of constructing a typical multi-level sampler used at Site 24 is provided in Table 3-III.

Table 3-III: A Breakdown of the Costs of Constructing a Typical MLS Used at Site 24 (1990 $).

Materials Costs:	
Aluminum tubing, 6061 grade, .035" wall, 3/16 O.D., 212 ft/MLS, $0.27/ft	$54.24
Swagelok© brass elbows, 8/MLS, $4.00 ea	$32.00
Swagelok© brass unions, 16/MLS, $2.20 ea	$39.60
Stainless steel screen, 200 mesh, 1 sq. ft.(sf)/MLS, $8.35 sf	$8.35
Aluminum porch screen, 20 mesh, 1 sf/MLS, $0.35 sf	$0.35
Hose clamps, size 36, all stainless, 16/MLS, $0.70 ea	$11.20
Glass wool, 1/4 pack/MLS (about 1/4 sf), $13.00 ea	$3.25
Machining Swagelok© caps into nuts	$10.00
Cleaning supplies (acetone, distilled water, towels)	$5.00
Total Materials Costs per MLS	$167.00
Labor Time:	
Cleaning materials	1/2 hour
In-house construction (drilling and making portholes)	1 hour
On-site construction(screening, joining PVC, Al tubing)	1 1/2 hours
Total Construction Time per MLS	3 hours

3.4.5 Depth of the Water Table Below Ground

The distance between the ground surface and the water table is important to the design of the MLS because it affects the sampling options available. The peristaltic pump is not able to apply suction to water that is over 25 feet (or about 8 meters) away, because it can only pull with one atmosphere of pressure. One atmosphere is equivalent to 33 feet of pressure elevation and minus friction losses, comes to about 25 feet. In such situations a bundle piezometer may be used with tubes wide enough to allow a narrow submersible pump or a mini-bailer, a device custom-made for use at the Borden site.

The "positive-displacement multilevel point-sampler" was developed for use at the Borden site specifically where the water table is more than 25 feet below ground (Cherry et al., 1983). In this MLS, small pump aids were designed to be located at each sampling port. A schematic diagram of this MLS along with a detail of the sampling port is shown in Figure 3-7. Sampling is accomplished by pulling the syringe plunger up with a hand pump at the ground surface (pressure in the aquifer usually causes the syringe to fill naturally) and pumping out to force the syringe plunger down, with check valves to direct the groundwater sample out of the sampler.

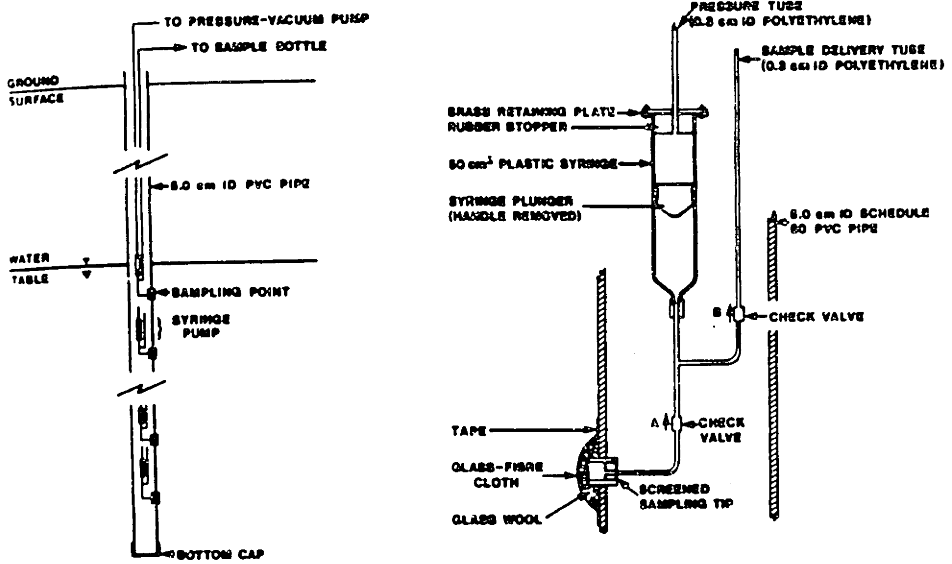


Figure 3-7: The "positive-displacement multilevel point-sampler" proposed by Gilham and Johnson and used at the Borden site (Cherry et al., 1983).

3.4.6 Installation Technique, Ease of Construction and Weight of MLS

Practical considerations may affect the MLS design, such as how much time and space is available to build and install the samplers. Because sampling wells are quite long, a certain amount of MLS assembly must be done on-site. If the samplers are simple to construct, less time will be spent waiting for the wells to be completed between drilling holes. If MLS construction is complicated, extra time is available, and the site being worked on is secure, the samplers can be constructed ahead of time and stored on-site. Of course, it is much easier to maneuver the MLS for installation if the sampler materials are light and rigid. One reason aluminum tubing was chosen over stainless steel for use on our MLSs was to save on weight, allowing the samplers to be easily carried.

The techniques available for installation of the multi-level samplers will affect how they may be designed. The inside diameter of the hollow stem auger or driven casing will dictate the maximum outside diameter of the MLS. The hollow stem auger available for use at Site 24 had an inside diameter of 4 1/4 inches, so we were limited to a 2 inch outside diameter PVC casing, allowing room for the sampling port coverings and the screws that hold the auger flights together.

3.5 Construction of Site 24 MLSs

Initial construction of the MLSs was carried out in the Parsons Lab work room. For the first step, all the exterior and interior metal surfaces were rinsed with acetone and then distilled water, to clean the metal of possible cutting oils. The Swagelok© elbows are each fastened to a 12 foot length of aluminum tubing. Holes are drilled into the 10 foot lengths of PVC pipe where each sampling port will be, just large enough to allow the threaded end of the elbow to fall through. The elbows are put into the holes by laying the PVC pipe horizontally, hole-down, and sliding the aluminum tubing elbow-first into the pipe, threaded end-down, until the threaded end of the elbow falls into the hole. The nut is then fastened to the threaded end on the outside of the PVC pipe, forming the sample port (see Figure 3-6 for sampling port detail). As each port is installed, its order along the sampler is labeled to keep track its depth during construction. The screening materials are placed over each port and fastened above and below with hose clamps. At this point the well sections and tubing are transported to the site in a 12 foot deep cargo van.

On-site, just prior to installation, the aluminum tubes are extended with Swagelok© unions and the PVC pipes are threaded together to bring each MLS to its full length. It is very important during this final step to maintain the label of each tube, so that after installation it is known which tube is connected to which port under the ground. Labels must be written and applied in a manner that is permanent. Our MLSs are labeled with tape and waterproof magic marker.

3.6 Siting the MLSs at Site 24

The positions on Site 24 where the MLSs were installed were determined using our best judgment. Criteria considered included where the naphthalene concentrations would be high, where the extent of contamination was great, where the data would be most informative (i.e. where little information was known) and very importantly, where the site had been cleared of trees so that it could be accessed by a drill rig. Much of the site was not accessible due to very thick shrub and tree growth. All the concentration and head data available at the time were considered in siting the MLSs. MLSs were sometimes installed near monitoring wells and where soil borings had been done in order to compare different types of data. Figure 3-8, Figure 3-9 and Figure 3-10 show the locations of the MLSs and monitoring wells on the site after installation of Rounds 1, 2, and 3 respectively (new MLSs are indicated by shading). These three maps were drafted by hand and are only meant to show the approximate locations of the well stations and MLSs on Site 24.

3.7 Installation Techniques

Techniques used for the installation of multi-level samplers are borrowed from those used for the installation of conventional monitoring wells. The most commonly used drilling method for MLS installation is hollow-stem augering, with the drive and wash technique somewhat less utilized. A hollow-stem auger is a hollow steel pipe with large threads on the outside. The auger is turned into the ground by the drill rig, flight by flight (an auger flight is 5 feet long), to the depth at which the MLS is to sit in the aquifer. The MLS is hoisted up over the top of the hollow auger and then dropped down inside. Once it is secure, the hollow auger is screwed back out of the ground. Securing the MLS involves waiting until the aquifer caves in around the bottom of the sampler, or filling up the MLS with clean water, so it is not buoyant as the groundwater fills up inside the hole. In drive

Figure 3-8: Site 24 after the installation of Round 1 MLSs, 4/10/89

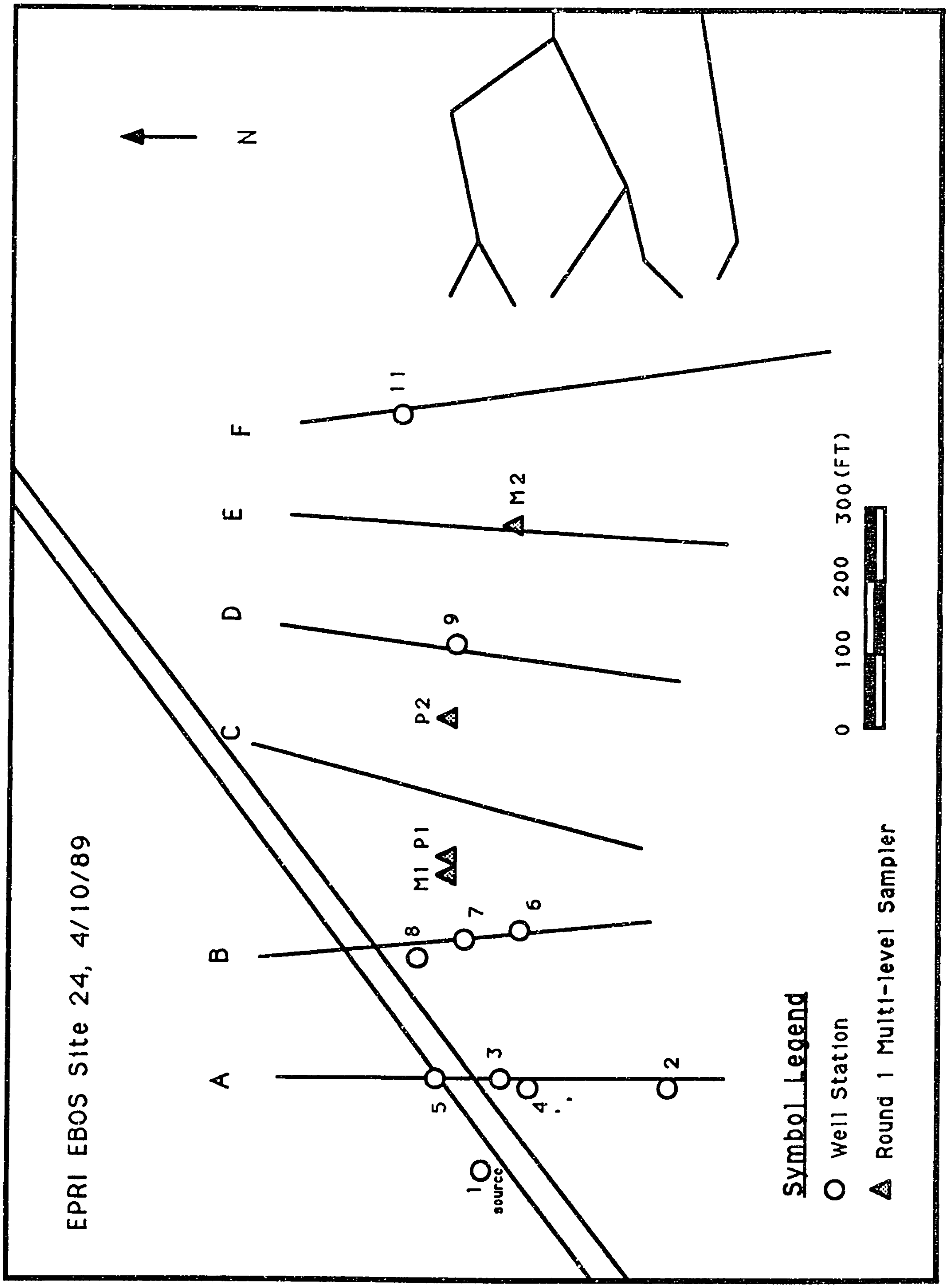


Figure 3-9: Site 24 after the installation of Round 2 MLSs, 9/13/89

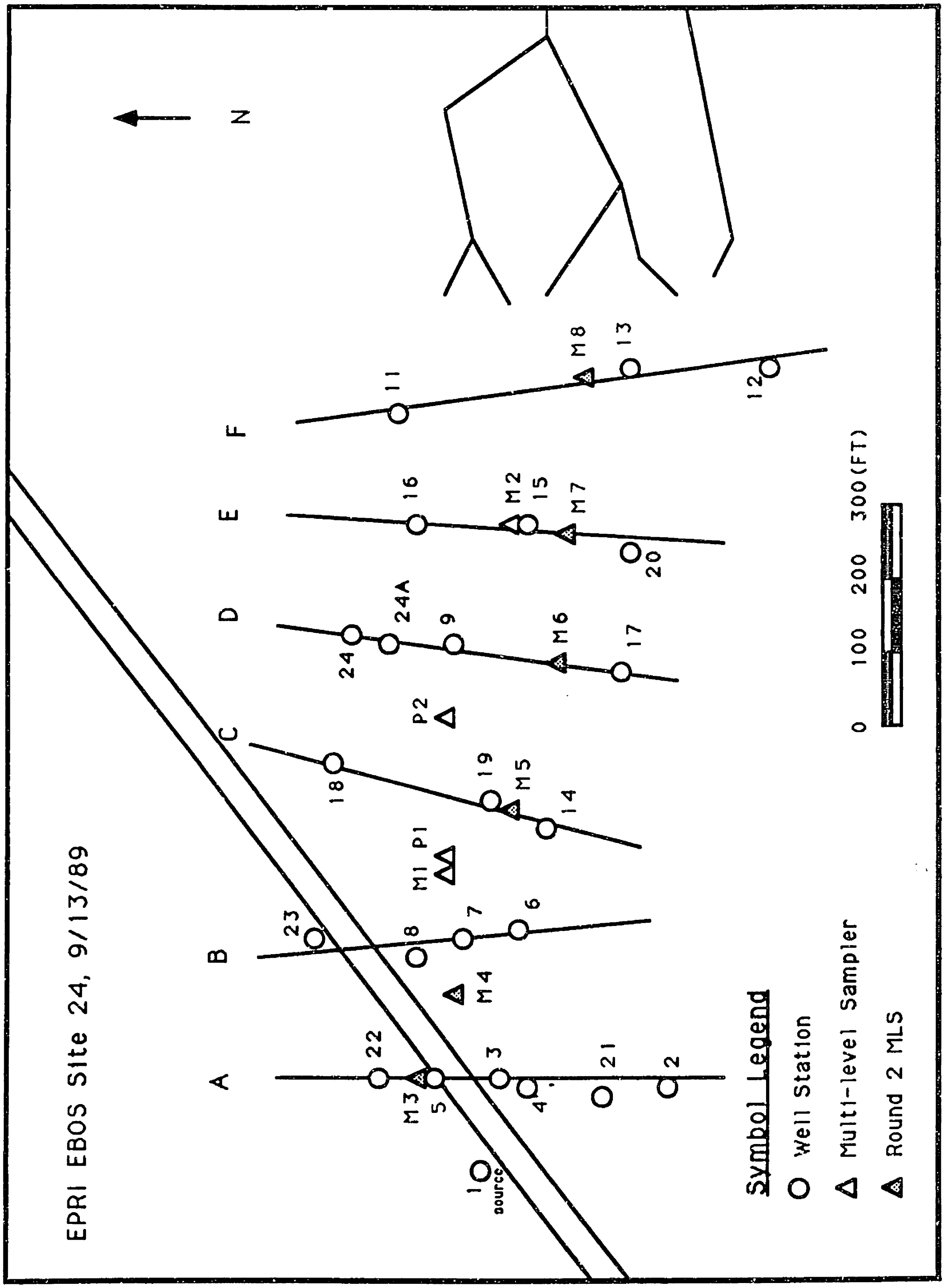
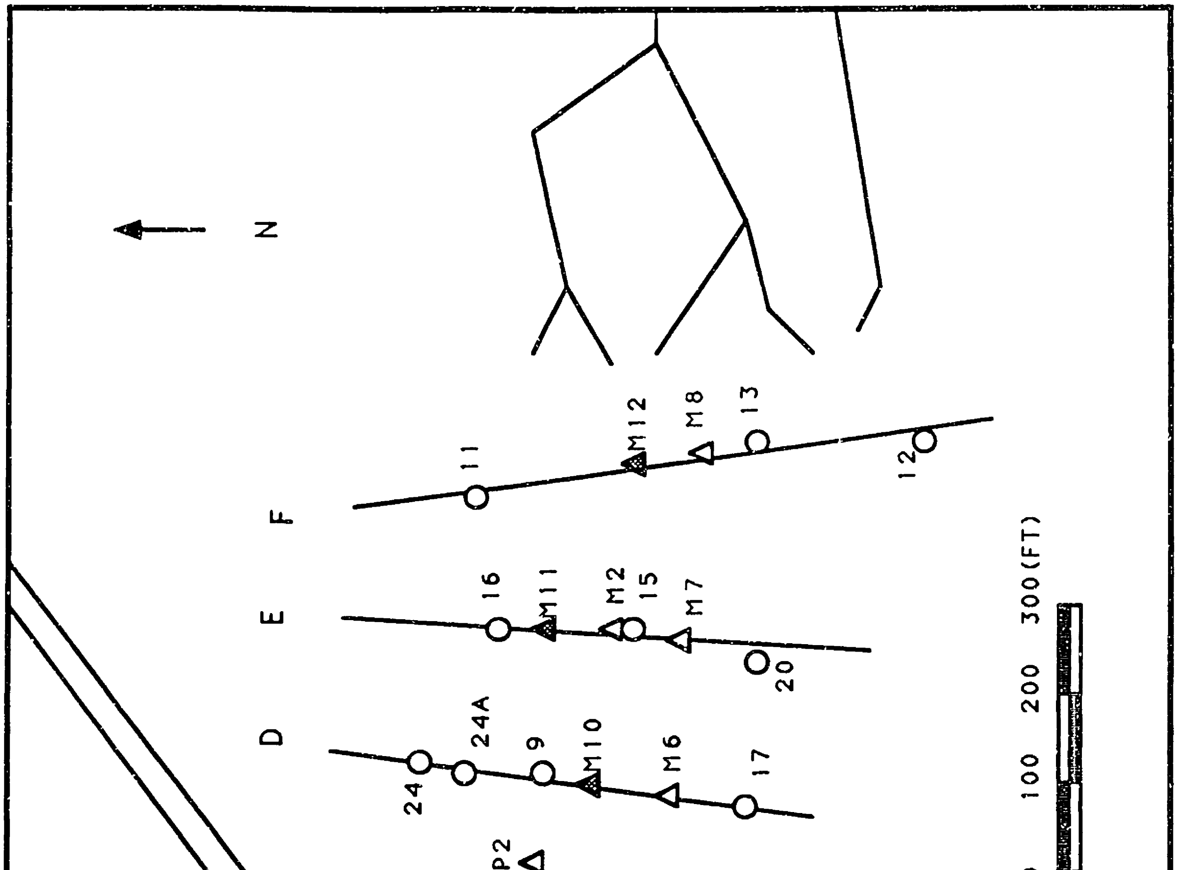


Figure 3-10: Site 24 after the installation of Round 3 MLSs, 4/3/90



and wash drilling, a steel pipe is driven into the ground by the drill rig and the soil inside of it is forced out by pressurized water. Hollow stem augering is preferred to the drive and wash method because it is faster and does not produce large amounts of leftover water and soil. If a site is heavily polluted, regulations often mandate the proper disposal of all materials removed from the site, and so disposing of the hazardous soil and water leftover from drive and wash drilling would be costly. The advantage to the drive and wash technique is that less aquifer material around the MLS is disturbed by the process.

After the MLS is installed and the auger or casing is removed, the empty annulus around the MLS must be filled. In this space clean sand or auger cuttings (the soil brought up by the auger threads) can be placed around the MLS and packers can be placed at specific depths to isolate zones of the aquifer along the well.

As part of the research of the large tracer test projects at the Columbus and Cape sites, various techniques for installation of MLSs were compared At the Cape, a test was carried out that compared tracer test results for a MLS installed four different ways: (1) hollow-stem auger, (2) hollow-stem auger with bentonite packs between each sampling port, (3) drive and wash, and (4) drive and wash with bentonite packs. The results showed "no apparent difference in the performance of differing MLS due to... installation methods in this type of aquifer." (Garabedian, 1987) Based on local tracer tests, the study at Columbus found that MLSs with bentonite packers between sampling ports installed by hollow-stem auger performed better than MLSs installed by hollow-stem auger (no packers) and drive and wash drilling (TVA, 1988).

3.7.1 Typical Installation of Site 24 MLSs

A hollow-stem auger was used to install our multi-level samplers for reasons of convenience; CAA had this type of drill rig on site for monitoring well installation. Constraints on drilling time prevented us from considering packing options. The unconsolidated soil in Site 24 readily collapsed around the samplers below the water table. The collapsing soil allowed the auger to be easily pulled up from around the MLS without disturbing the placement of the MLS. When the hole was drilled through a clay layer, as it was in Round 2, the soil would not collapse, but the hole would fill up with water. In these cases the leak-free MLS became buoyant and several times had to be filled with fresh water (stored on the drill rig) in order to keep it from floating. After the auger was removed, the area around the MLS was backfilled with sandy soils brought up by the auger. An effort was made during installation to backfill with the uncontaminated sandy soils from the top elevations. The proceeding Section 4.3, "Tracer Tests: Detection of Improper Soil Collapse Around the MLSs" describes in situ tests carried out to assess whether backfilling with sandy auger cuttings was sufficient to create a proper MLS/aquifer interface.

Round 1 and 2 MLSs were all installed to a depth set by estimating the location of the clay layer by soil stratification charts drawn up by AES. These charts were drawn from a limited amount of information on soil types recorded during soil borings. Since more time was available during the installation of Round 3 MLSs, split spoons were used to locate the precise depth of the clay layer, and thus the MLSs could be situated as close to the clay without going into it.

Protective metal well casings with lockable lids were cemented into the ground over each sampler, as the final step of installation. The cement seal serves to prevent surface infiltration, as well as fasten the casing to the ground. Installation time is approximately one hour per MLS if clay is not encountered, and two hours with clay. The extremely viscous clay delayed installation because it gripped the auger and made the auger flights difficult to remove.

After installation in Site 24, we developed the MLSs. This is done to remove any fine soil particles that might have been stirred up during drilling that could clog the narrow tubes of the MLS if allowed to sit. MLS development involved pumping on every port for at least an hour at about 60-70 ml/min.

3.7.2 Special Installation of MLS M-12 Through a Clay Lens

Special care was taken to install MLS M-12 on transect F. At this location, about 25 feet below the ground surface, a clay lens exists. Because data from soil borings indicated the presence of naphthalene contamination below this clay layer, we wanted to install a MLS that would be able to take groundwater samples from below and above this clay lens while maintaining the isolation of these two areas. This was accomplished by grouting around the MLS at the depth of the clay lens.

Installation of M-12 took all day (rain didn't help either) and involved many steps. As the auger was drilled down, split spoons were used to pinpoint the depth of the top and the bottom of the clay lens and the depth of the lowest layer of clay. When the hollow stem auger was drilled down to the where the bottom of the MLS was to rest, the MLS was dropped into the auger. The auger was then pulled up to be even with the bottom of the clay lens, and the area around the MLS that had not caved was backfilled with coarse sand by dropping it down into the auger (around the MLS). At this point a slurry of bentonite was prepared by the drillers in a large barrel. The bentonite was transported to the bottom of the hollow stem auger via a "tremmie pipe", a pump with thin steel pipe attached to the end of a very long hose. Just enough bentonite was pumped into the auger as would fill up the space around the MLS once the auger was removed, and not go beyond the height of the clay lens. This volume was checked before the auger was raised by checking the height of the bentonite in the auger and doing a quick calculation. Little time could be spared to pump in the exact amount of bentonite, however, because of the threat of the bentonite hardening and sealing the MLS into the auger. After an adequate amount of bentonite was put into the auger, the auger was brought up and the rest of the annulus backfilled with coarse sand to above the sampling ports, and then sandy aquifer material brought up by the auger. Figure 3-11 shows the final subsurface setting for MLS M-12.

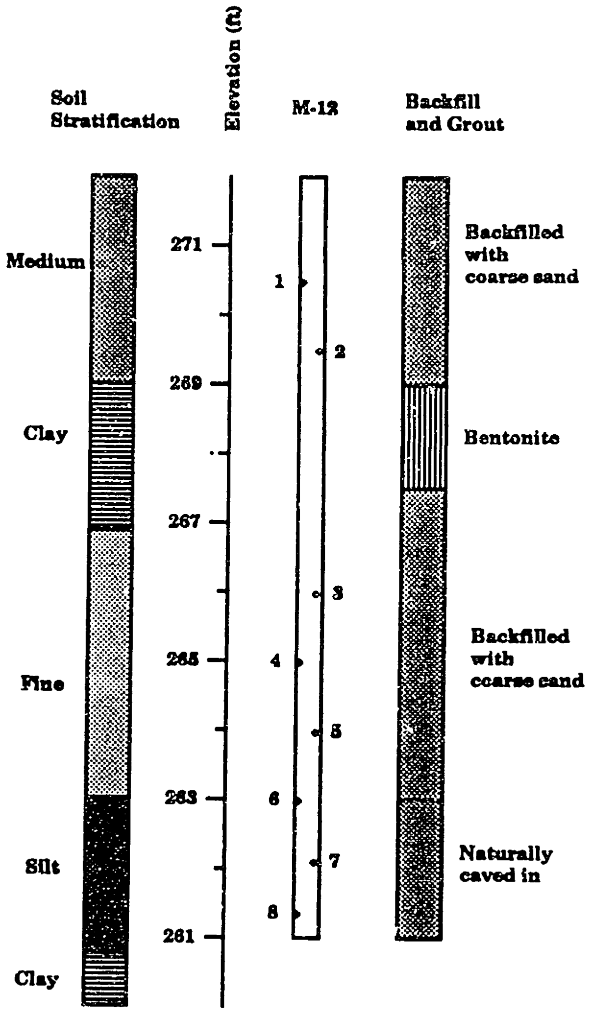


Figure 3-11: Special installation of MLS M-12 and corresponding soil layers.

3.8 Sampling the MLS

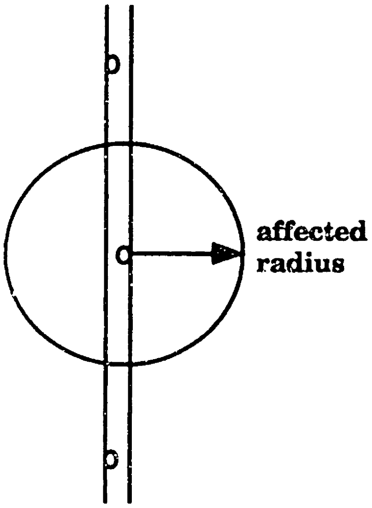
Groundwater sampling is done at pumping rates that are as slow as possible. Slow rates come closest to simulating actual groundwater flow velocities and minimize turbulent flow that can stir up fine (non-mobile) particles.

Improper sampling protocols may introduce large sources of error. In order to preserve the groundwater's chemistry after it has been withdrawn from the aquifer, care must be taken to avoid its contact with improper or unclean materials (see previous section entitled Contaminant (Materials) Considerations). Groundwater samples, once in the vials, must contain as little air (head space) as possible; the presence of a second medium could alter the chemistry of the solute. They must also be stored on ice to minimize microbiological activity in the groundwater sample. Finally, solute samples must be preserved and then analyzed within certain time periods depending on the stability of the contaminant to be analyzed for. These and others are all standard steps prescribed by the U.S. EPA that must be taken when doing groundwater sampling by any technique.

3.8.1 MLS Sampling at Site 24

Sampling is carried out on MLSs at least a couple weeks after installation and development. This period of time is set to allow the groundwater and the chemicals in it to redevelop equilibrium conditions in the disturbed areas around the MLSs. For a certain period prior to sampling a MLS, all its ports are pumped on (purged) simultaneously. In conventional well sampling purging is done to clear the well of stagnant water, but this is accomplished in the MLS within a minute, because the volume of stagnant water in the narrow sampling tubes is so small. Purging of MLSs is done to develop horizontal flow patterns and is done in an effort to draw groundwater for sampling from beyond the area around the MLS disturbed during installation. In other words, the groundwater taken for the sample is pumped from a radius equal or greater than the extreme outside radius of the auger. Figure 3-12 shows the radius calculated from the time and rate of purging carried out before sampling at Site 24. Purging done at a minimum of 60 ml/min for 20 min affected a radius of 9.2 cm, which equals the radius affected by auger installation.

A special sampling train was developed for MLS sampling and can be seen in Figure 3-13. Cole-Parmer peristaltic pumps were used, and all ports were sampled simultaneously.



Ave. time of pumping before sampling (t) = 20 mins

Ave. pump flow rate (Q) = 60 ml/min

Porosity (n) = 0.35

Internal vol. (I.V.) of 25 ft sample tubing (r = 1.3 mm) = 40.5 cu. cm

Vol. affected x porosity = Vol. water removed - Internal vol. of sample tubing

4/3 x pi x radius ^{3} x n = Q x t - I.V.

Affected radius = cube root[3 x(Q x t - I.V.)/(n x 4 x pi)]

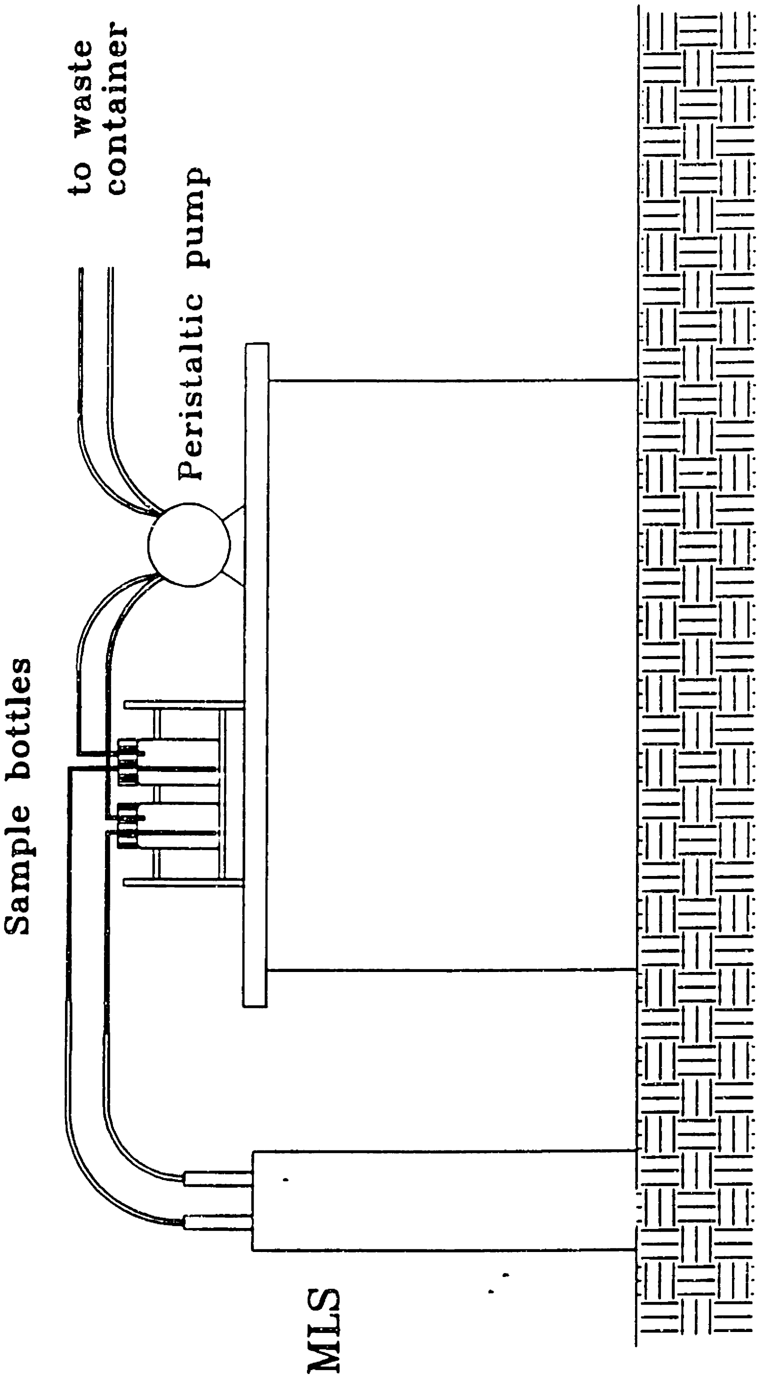
Affected radius = 9.2 cm

Disturbed radius = auger outside radius = 3 5/8 in = 9.2 cm

Figure 3-12: A comparison of radius of aquifer affected during pre-sampling pumping with radius of soil disturbed by auger installation of MLS.

A very slow pump rate of 30 ml/min was used for sampling. Once horizontal flow was established and the sampler tubes were adequately purged, the withdrawn water was directed from each tube into a lab-cleaned 40 ml borosilicate glass sample vial. A diagram of the sample vial set-up used is shown in Figure 3-14. Aluminum tubes, connected to the MLS tubes with short plastic tube connectors, bend into the sample vial through holes punched in their teflon-coated septum. The groundwater is emptied smoothly onto the bottom of the vial to avoid volatilization from aeration. The air in the vial is withdrawn through an aluminum tube that exits the top of the vial through the septum, by the action of the peristaltic pump. The pump is placed after the sample vial in the sampling train because only flexible silicone tubing may be used in the pump heads and these may be adsorbent to organic compounds. Because the peristaltic pump uses a slight vacuum to draw up the water, degassing of volatile compounds from the groundwater is a potential concern. Since we are only concerned with naphthalene which is considered only semivolatile, degassing is not expected to be a problem.

Figure 3-13: The sampling set-up used for taking groundwater from MLSs at Site 24



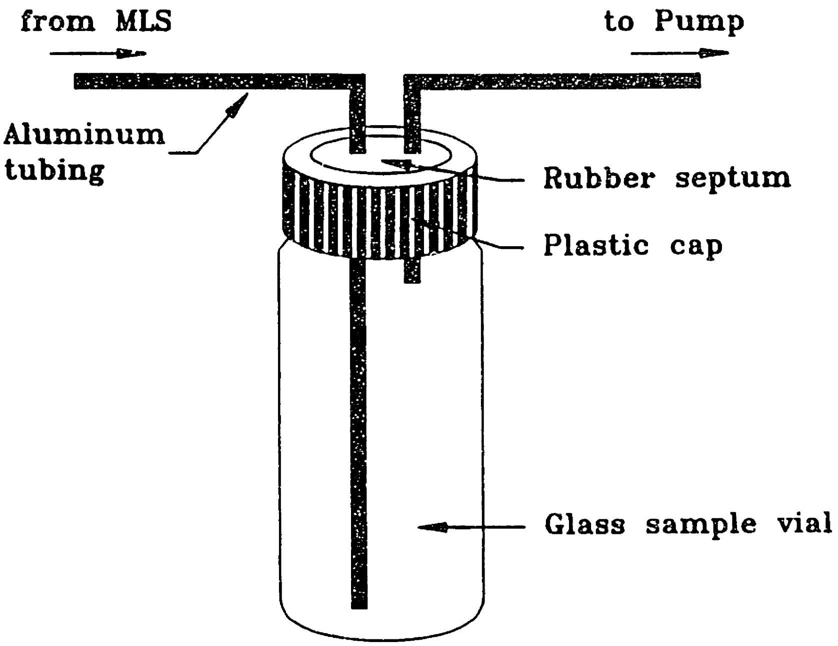


Figure 3-14: The vial used for collecting groundwater from MLSs at Site 24

3.8.2 Sampling Equipment Effects

A number of studies have been done to test the effects of sampling equipment selection on the concentration of organic compounds in groundwater samples. The effects have to do primarily with materials in the apparatus that come in contact with the groundwater sample, and aeration and pressure changes, which can induce volatilization. In one study, three submersible pumps, a centrifugal pump, two peristaltic pumps and a bailer were used to take trichloroethylene (TCE) samples from a monitoring well (Pearsall and Eckhardt, 1987). In this study Pearsall and Eckhardt found that the peristaltic pump with silicone tubing had significantly lower TCE concentrations than the samples from the submersible and centrifugal pumps or the bailer. The silicone tuning was responsible for this loss since the use of a peristaltic pump with Teflon tubing (except in the pump heads) did not have as pronounced an effect. They concluded, however, that "pump placement, rate of pumping, duration of pumping, and the uniformity of the vertical and lateral distribution of TCE in groundwater near the well screen ... can have a greater influence than the type of sampler used". In a laboratory experiment used in another study, three volatile organic compounds at low concentrations were sampled from a monitoring well set in a tank of mixed solution only (no medium) (Schalla et al., 1988). No statistical difference was found among four sampling systems used: a stainless steel and Teflon piston pump, a Teflon bailer, a Teflon bladder pump, and a PVC air-lift pump.

A study was done to investigate biases in sampling the bundle piezometers described by Cherry et al. in GWMR, 1982 (Barker et al., 1987). The results of the study indicated that potential biases due to volatilization, sorption and leaching are minimal or can be easily controlled in most situations. Barker et al. also found one particularly unusual result, that organic compounds can penetrate through polyethylene (PET) tubing. Transmission through PET tubing can have implications if a very long length of sampling tubing is set in standing well water that is more contaminated than the groundwater. Teflon tubing did not allow the transmission of organics, and "centuries are required" for transmission through PVC piezometer pipe. Another important finding by Barker et al. was that organics sorbed to tubing materials from previous sampling can be released in subsequent sampling events. Both Teflon and polyethylene exhibited this desorption behavior. In general, the studies indicate that careful selection of the appropriate sampling materials and techniques depends on the sorptive and volatile properties of the organic compound. Proper precautions can minimize sources of uncertainty involved in the collection of groundwater samples.

3.8.3 Collection of Head Data from MLS

It is possible to measure the water levels in the thin tubes used for sampling in a multi-level sampler. Pickens et al. describe a device for measuring hydraulic head from a cased multi-level sampler, but it is only accurate where piezometric levels are close to the ground surface (by several meters) (Pickens et al., 1978). For the Columbus site project, a vacuum manifold device was developed to allow the measurement of the hydraulic heads in up to nine sampling tubes (from one MLS) simultaneously (TVA, 1988). Vertical hydraulic head gradients were calculated and used to compare samplers of different design; a good correlation was found between the vertical hydraulic head distributions in the augered MLS with packers and the driven stainless steel MLS. The average difference in the mean vertical hydraulic gradient measured in adjacent but different samplers was six percent, indicating that their vacuum manifold device gives fairly reproducable results. Hydraulic head measurements were not collected from MLSs at Site 24, because the procedure is known to be inconvenient and has poor accuracy (Cherry et al., 1983).

Chapter 4

Site 24 Results

The final stage of this project is the collection of groundwater samples from the MLSs and interpretation of the naphthalene concentrations found in those samples. Three rounds of samplers were installed, and four sampling rounds were carried out over the period from the spring of 1989 to the spring of 1990. The MLSs were named M or P depending on if the sample tubes were PVC or metal (aluminum) and numbered according to their order of installation.

All groundwater samples from the MLSs were collected by me with one assistant. Students from the Civil Engineering Department who assisted me at one time or another were undergraduate Sharon Chern and Daniele Agostini and graduate students Lynn B. Reid and Roland Springer. The samples were preserved by extraction by employees of CAA or by me, and all were analyzed by CAA on site or in Boston at CAA's accredited laboratory. All the data on the concentrations of naphthalene in the groundwater collected from Site 24 MLSs in four sampling rounds are provided in Appendix A.

With the solubility of naphthalene in water at about 88 ppm (see Table 2-III), and the mole fraction of naphthalene in the tar at Site 24 approximately 0.053, the predicted aqueous concentration of naphthalene should be about 4.7 ppm (Groher, 1990). With only a few exceptions, all the results show naphthalene concentrations less then this value. Concentrations above this value are probably due to the presence of droplets of undissolved tar in the sample.

4.1 Trends in Data Variation

In order to more accurately interpret the MLS data, it is useful to first pinpoint the sources of data variation. Major differences in steps taken to acquire the data can result in errors. In this project, different MLS materials and analytical techniques were used. Natural variations such as recharge and groundwater flow can also cause data taken from the same point to vary at different times.

4.1.1 Representative MLS Data

The MLS should give fairly discrete values of a contaminant's concentration in the groundwater in the vertical direction. This contrasts with a monitoring well, which gives an average of a chemical's concentration over the depth of the well screen (see Section 3.2, "Data Collection from a MLS Versus a Monitoring Well"). Representative naphthalene data taken from samplers P-1, M-1 and M-5 are shown in Figure 4-1. The variation of concentration values along the length of each MLS displayed in this figure indicates that the MLS is taking discrete samples. If the samples were not from discrete zones, the values would be closer in value or averaged to some degree.

4.1.2 Analytical Technique Sensitivity

Almost all of the groundwater samples taken from MLSs at Site 24 were analyzed by CAA by gas chromatograph with a flame ionization detector (GC/FID). A few of the samples were also analyzed by gas chromatograph with a photoionization detector (GC/PID), for purposes of comparison. A third technique used very often for analysis of water and soil samples is gas chromatograph with mass spectroscopy (GC/MS). This machine was used to analyze some of the monitoring well data taken by CAA. It is important to consider the sensitivity of the analytical technique used to know if the concentrations being detected are accurate and consistent.

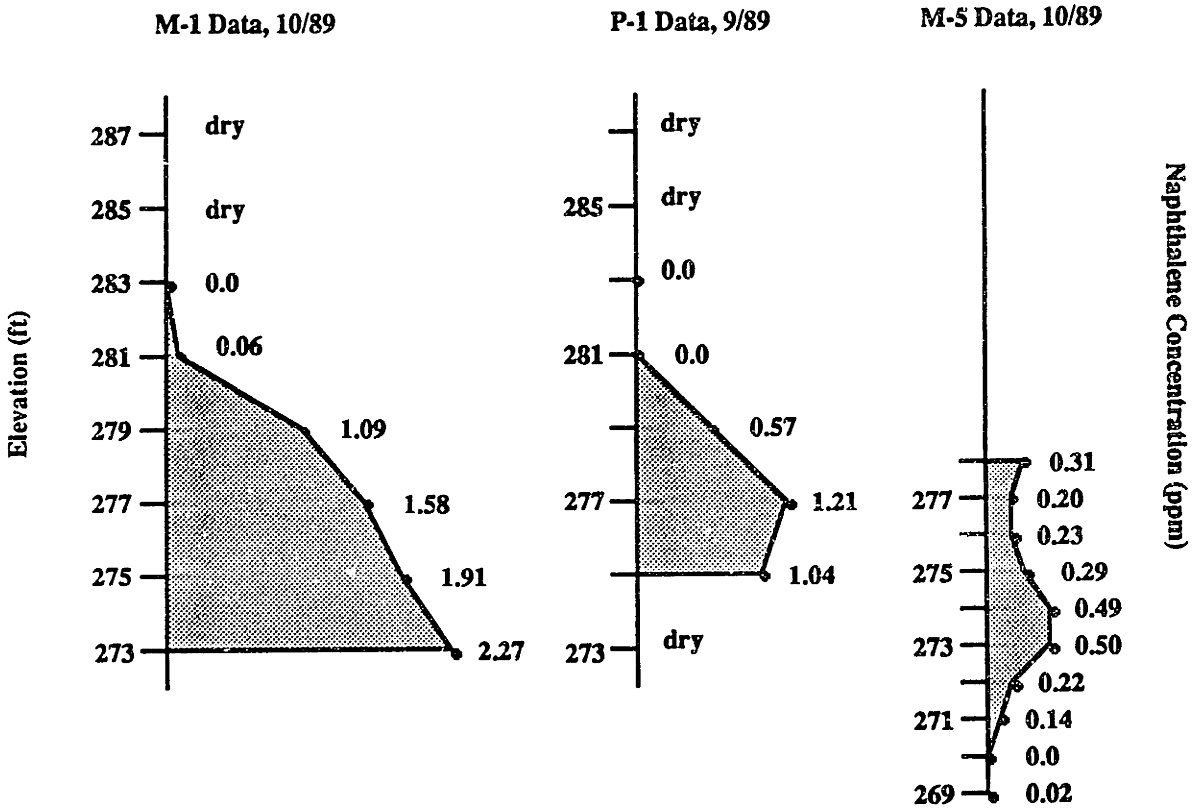


Figure 4-1: Typical MLS data profiles.

These gas chromatographs basically work by first vaporizing the chemicals, and then accelerating the excited molecules towards a detector. The molecules are separated by their weight since the lightest molecules will reach the detector first. The output of the detector is in the form of energy versus time. From the order of the chromatograms (from lowest to highest molecular weight) and the intensity of the peaks relative to standards, chemists can calculate the concentration of chemical constituents sought.

These three techniques are not equally suited for analysis of organic chemicals. A photoionization detector, in using light to ionize the sample, has a better ionization efficiency than a flame ionizer, which can volatilize and thus lose some of the organic content of the sample. This effect can be seen in a comparison of MLS data analyzed by GC/FID and GC/PID, shown in Table 4-I. The GC/FID, from the average of four samples, detected concentrations 14 % lower than GC/PID for the same samples.

Table 4-I: A Comparison of MLS Data Analyzed by GC/FID and GC/PID for Naphthalene (in ppb)

Sample ID	GC/PID	GC/FID	% Lower
P-1-14	800	660	17.5
P-1-16	1070	895	16.4
P-2-20	1510	1410	6.6
M-1-14	1840	1530	16.9
Average			14.3% lower

Gas chromatograph with mass spectroscopy (GC/MS) is a technique often employed by analytical laboratories for its sensitivity to a wide range of compounds. While GC/MS was not used in this project to analyze groundwater samples taken from the MLSs, it is important to emphasize the variations associated with different analytical techniques. Table 4-II shows a comparison between Site 24 soil samples analyzed by GC/FID (a portable field unit) and by GC/MS (permanent lab unit). Table 4-III shows the variation of naphthalene concentrations in groundwater samples taken from Site 24 monitoring wells analyzed by GC/FID and GC/MS. The GC/MS is the same as the one cited in Table 4-II but the GC/FID is a different unit, permanently located in CAA's Boston laboratory like the GC/MS.

These two tables comparing GC/MS data with data from two different GC/FID units brings out two important points about the way data can vary with the analytical technique. The first point is the same just described in the comparison between GC/FID and GC/PID: different machines have different sensitivities. The same samples had averages of 18% and 105% higher concentrations when analyzed by GC/FID as compared to GC/MS. The second important consideration to make is who did the analyses. The use of the field/portable GC/FID resulted in much higher concentrations than the lab unit (as they were both compared to GC/MS). This is because the analytical lab who did the analysis (CAA) is organized in a production-line fashion. Since the laboratory must regularly perform a large volume of analyses, the workers there do not look over the chromatograms with great deliberation. The chromatograms produced in the field were looked over several times and interpreted with great care, so peaks that might have been overlooked by the lab were noticed by the chemists in the field.

Table 4-II: A Comparison of Site 24 Soil Data Analyzed by GC/FID (Field Unit) and GC/MS for Naphthalene (in ppb)

Sample ID	GC/MS	GC/FID	% Higher
B1-1-1	50	50	0.00
B-1-2	80	480	500.00
B1-1-3	300	500	66.67
B1-2-1	100	50	-50.00
B2-2-2	8900	13000	46.07
B2-2-3	480	1500	212.50
B2-3-1	110	260	136.36
B2-3-1D	140	190	35.71
B2-3-2	0	0	0.00
Average			105.26%

Table 4-III: A Comparison of Site 24 Monitoring Well Data Analyzed by GC/MS and GC/FID for Naphthalene (in ppm)

Sample ID	GC/MS	GC/FID	% Higher
MW-2	3.3	4.307	30.52
MW-2D	0.82	0.854	4.15
MW-2T	1.4	1.674	19.57
MW-6	0.061	0.061	0.00
MW-9	1.2	1.629	35.75
MW-9D	1.1	1.241	12.82
MW-9T	0.67	0.668	-0.30
MW-12	0.74	0.925	25.00
MW-12D	1.06	1.174	10.75
MW-12T	1.1	1.244	13.09
MW-13	3.1	4.251	37.13
MW-13D	2.6	3.337	28.35
Average			17.69%

In sum, these data indicate that GC/FID is more efficient than GC/MS for the analyses of naphthalene, but GC/PID is more sensitive than them both. Careful analysis by hand was possible for this project because of the small volume of samples involved, but this is not always an economical option. These figures show that it is important to analyze with the same technique, and preferably the most sensitive one (if economically possible), throughout a monitoring project, to maintain consistency of the data.

4.1.3 Variation of Duplicates

With a particular analytical technique there is always a degree of error associated with each measurement. To get a feeling for this accuracy, duplicate samples, spiked samples and blank samples are routinely analyzed as part of an analytical laboratory's quality assurance/quality control program. In sampling at Site 24, one duplicate sample was taken for every ten samples taken. Duplicate samples were taken from MLSs by sampling the MLS immediately after the first set of samples is taken. Even if only several of the MLS ports are samples for duplicates, the non-sampled ports are pumped to maintain typical sampling conditions. Duplicate samples that result in concentrations very close to the original samples taken confirm that standard sampling and analysis procedures were followed and also confirm the consistency of naphthalene in the groundwater around the MLS. Duplicate MLS samples that I took were analyzed by CAA by GC/FID. This data, shown in Table 4-IV, indicates that the margin of error associated with any concentration value taken from a MLS is about (plus or minus) 10%.

4.1.4 Effect of PVC Collection Tubing on Concentration Data

In Round 1, two different types of MLSs were constructed and installed, two with PVC collection tubing and two with aluminum. Plastic tubing was considered first because it was used in most previous designs (i.e. at Borden, at the Cape, and in Mississippi) where inexpensive materials were advantageous due to the large number of MLSs made.

Table 4-IV: Variation of Duplicates of MLS Samples Analyzed by GC/FID for Naphthalene (in ppb)

Sample ID	Dupe #1	Dupe #2	% From Ave.
	Jun-89		
P-1-16	920	870	2.79
M-1-14	1650	1410	7.84
	Sep-89		
P-2-20	182	241	13.95
	Oct-89		
M-3-7	2058	1958	2.49
M-3-8	1653	1101	20.04
M-3-9	1670	1665	0.15
M-3-10	1414	2034	17.98
M-4-8	754	762	0.53
M-4-9	439	453	1.57
M-4-10	563	559	0.36
	Apr-90		
M-11-3	1082	867	11.03
M-11-4	523	613	7.92
M-11-5	735	803	4.42
M-11-6	780	319	41.95
Average	Variation		9.50%

Aluminum was later considered for Site 24 MLSs due to the possibility of naphthalene adsorbing to the plastic. A PVC and aluminum MLS were installed a foot apart between transects B and C in order to compare the naphthalene concentrations they would generate.

Comparisons of groundwater samples I took on 6/89 and 9/89, analyzed by CAA by GC/FID (gas chromatograph with a flame ionization detector), are shown in Figure 4-2. These figures show that data from aluminum samplers is consistently higher than data from PVC samplers, in seven out of nine data points. This is most likely due to adsorption of naphthalene on the large PVC area the groundwater has to contact during sampling.

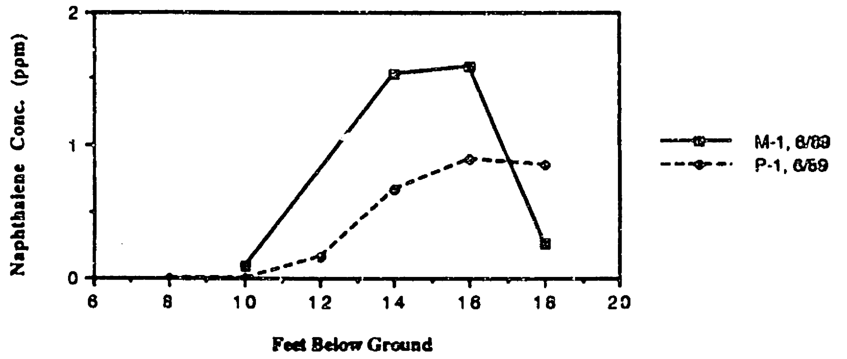
Because of these results, the MLSs built in subsequent rounds were built with aluminum tubes only. In addition, results from the P-1 and P-2 will not be used in comparison with the rest of the data.

An argument could be made for natural heterogeneity causing the discrepancy in values for MLSs that are placed a foot apart. It is conceivable that lower concentrations are naturally found a foot away due to variations in soil properties. At Site 24, however, the soil in most layers is fairly homogeneous. There is very little rock or organic matter to cause immediate changes in soil texture. Also, the duplicates confirm that different water taken from the same place at the same time has about the same naphthalene concentration. Because the trend of higher concentrations from the aluminum MLS held up over time, it is reasonable to infer that the difference in materials is responsible for the difference in concentration values. This topic does, however merit further study.

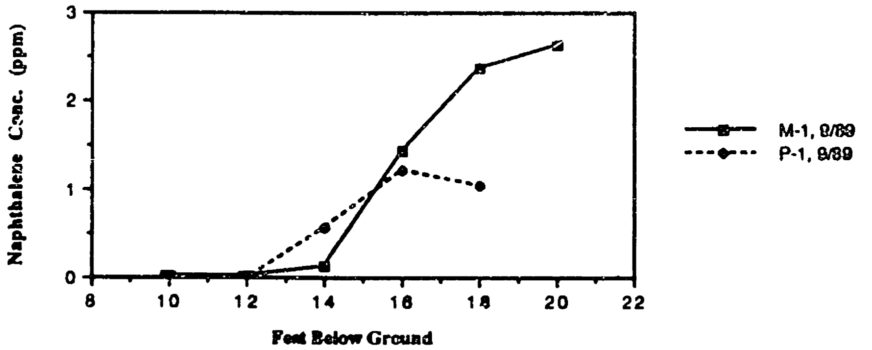
4.1.5 Variation of Data Over Time

Besides problems with sampling, materials, and installation, other sources of error exist that cause unpredictable variations in data taken from the same point. The concentration of a chemical constituent in groundwater can vary over time in the short-term due to a wide range of environmental factors, notably recharge. In other words, there is a certain margin of variation to be expected when comparing data taken at different dates. If this margin is too great, it might not be possible to make the comparison.

Variation of data over time can be studied by examining naphthalene concentrations taken at different dates from the same sampling port, by the same sampling technique, and analyzed by the same machine. Because the analysis of data is so expensive, only the samplers installed in Round 1 were sampled more than once. Data was also limited in the first sampling round because a number of sample bottles broke when they froze in a freezer that was set too cold. Figures 4-3 and 4-4 show the variation of MLS data with time. In



a) M-1 and P-1 Sampled 6/89



b) M-1 and P-1 Sampled 9/89

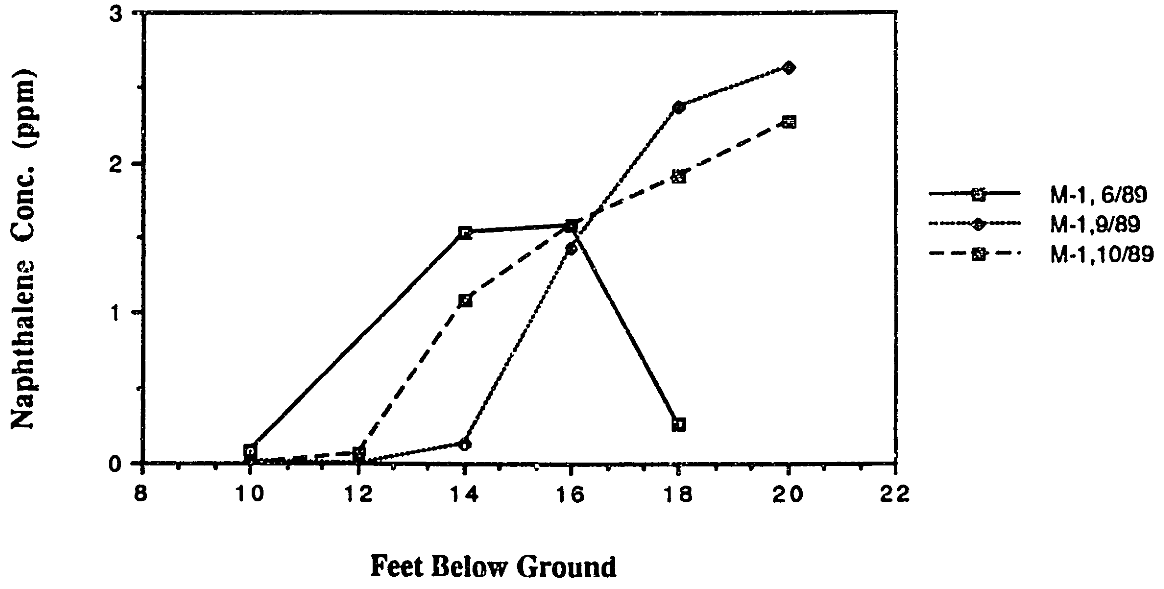
Figure 4-2: A Comparison of data from PVC and Aluminum MLSs.

some instances, data taken at the same point differed as much as an order of magnitude greater and smaller over a three month period (see M-1 data, ports 14 and 18 feet below ground; M-2 data, 21 feet below ground and P-2 data, 20 feet below ground).

The cause of such a wide difference in concentrations from the same point is difficult to determine. This could be explained if the whole plume had shifted downward vertically due to an extreme recharge event. Rainfall data for the area for the period of the three sampling rounds, displayed in Figure 4-5, does not seem to show any indication of a rainfall event that would cause the plume to be lower on September 5 and October 4 and 5 than on June 6, 1989. Such wide variation is seen in groundwater data taken from monitoring wells too. It is important then, where naphthalene is detected, to take more than one sample from

Figure 4-3: Variation of MLS M-1 and M-2 concentration data over time.

a) M-1 Data, Sampled 6/89, 9/89 and 10/89



b) M-2 Data, Sampled 6/89, 9/89 and 10/89

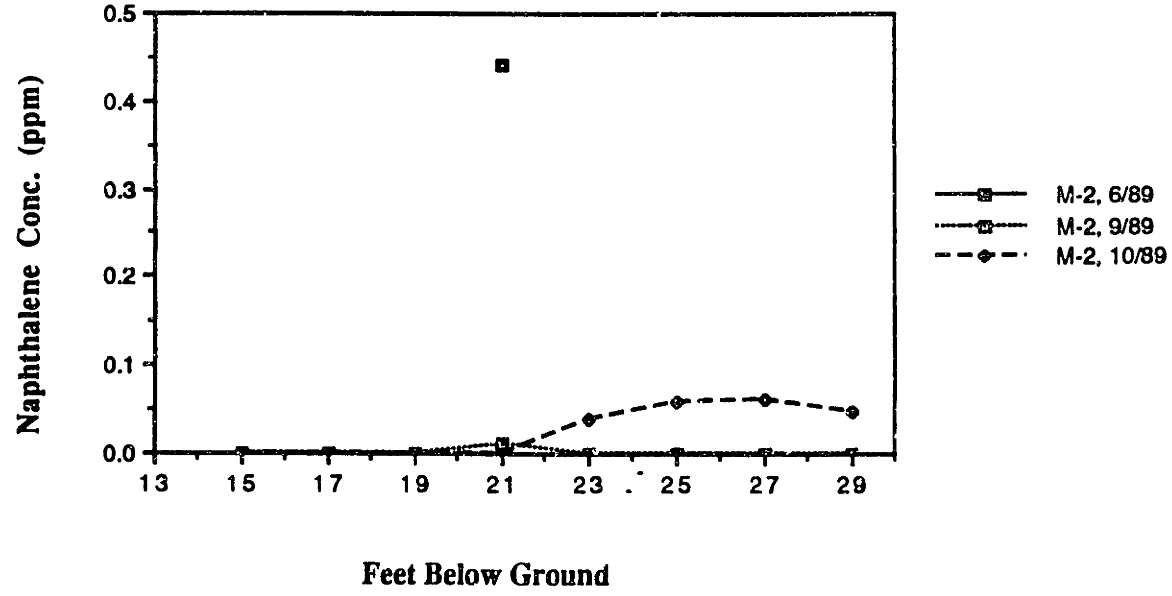
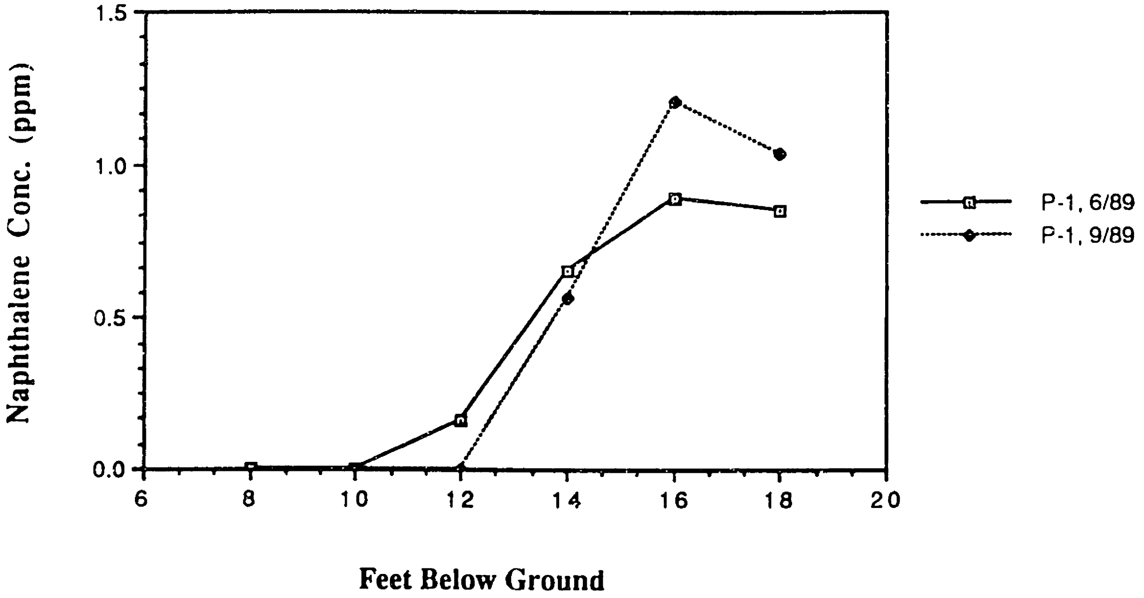
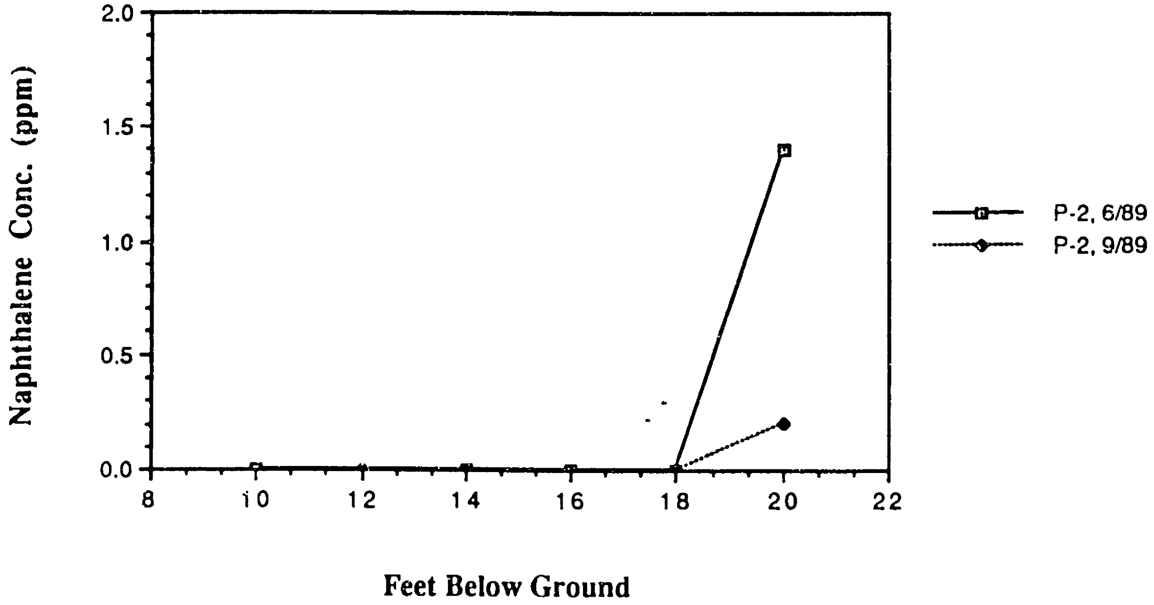


Figure 4-4: Variation of MLS P-1 and P-2 concentration data over time.

a) P-1 Data, Sampled 6/89 and 9/89



b) P-2 Data, Sampled 6/89 and 9/89



a point. Repeated sampling could show that one data point was an anomoly. Extreme errors in extraction of the contaminants from the sample medium prior to analysis could produce a bad data point. Concentrations that are very low (less than 0.1 ppm naphthalene) do not vary significantly over time, because the time variation seems to be based on a percentage of the concentration, not an absolute number.

4.1.6 Consideration of MLSs Installed Into the Clay

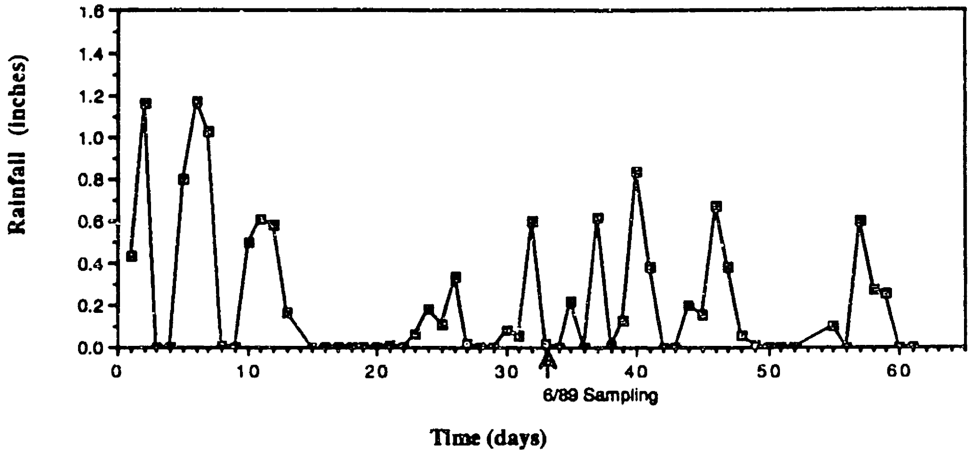
The four MLSs installed in Round 1 were installed to right above where the clay layer was expected to be. Based on soil data collected later by CAA that showed that naphthalene was present in the top of the clay layer, all the samplers installed in Round 2 intersected the clay. Another motivation for installing the samplers so deep was the hope that if several sampling ports were placed into the clay, the elevation of the lower edge of the plume could be identified (the plume could be bounded).

Figure 4-6 shows data profiles taken from two samplers installed into the clay next to the soil stratifications at those locations. The trend common to both data profiles indicates the basic problem with MLSs installed into the clay. The naphthalene concentrations from the ports within the clay area are about the same as the values from the ports just above the clay, when they are expected to be near zero. Since clay soils do not generally collapse around the MLSs, the annulus between the sampler and the intact aquifer material must be backfilled. In the Site 24 project the sandy material used for backfill is probably more conductive than the undisturbed clay outside the annulus. As a result, the contaminated water from above the clay probably filled in the area drilled out between the clay and the sampler to give the high concentrations found.

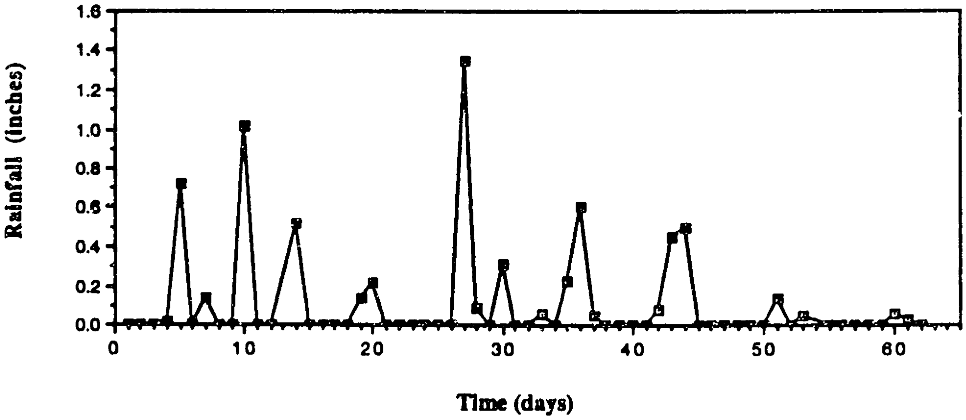
These problems suggest that the samples taken from the ports in the clay layer may be invalid, i.e. they may not represent the concentration of naphthalene in the clay. The presence of the break in the clay layer will not affect the flow of the groundwater above it,

Figure 4-5: Rainfall in the Site 24 area for the period May-October 1989.

a) Daily Rainfall (May, June 1989)



b) Daily Rainfall (July, August 1989)



c) Daily Rainfall (Sept., Oct. 1989)

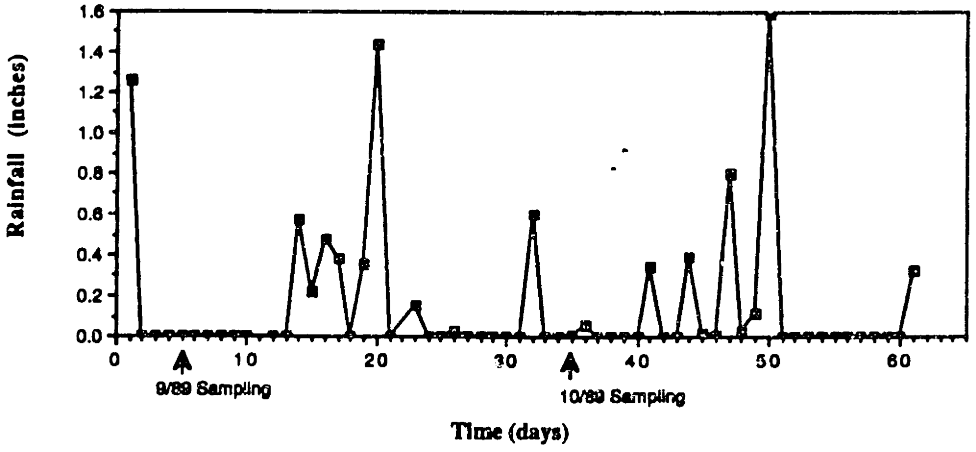
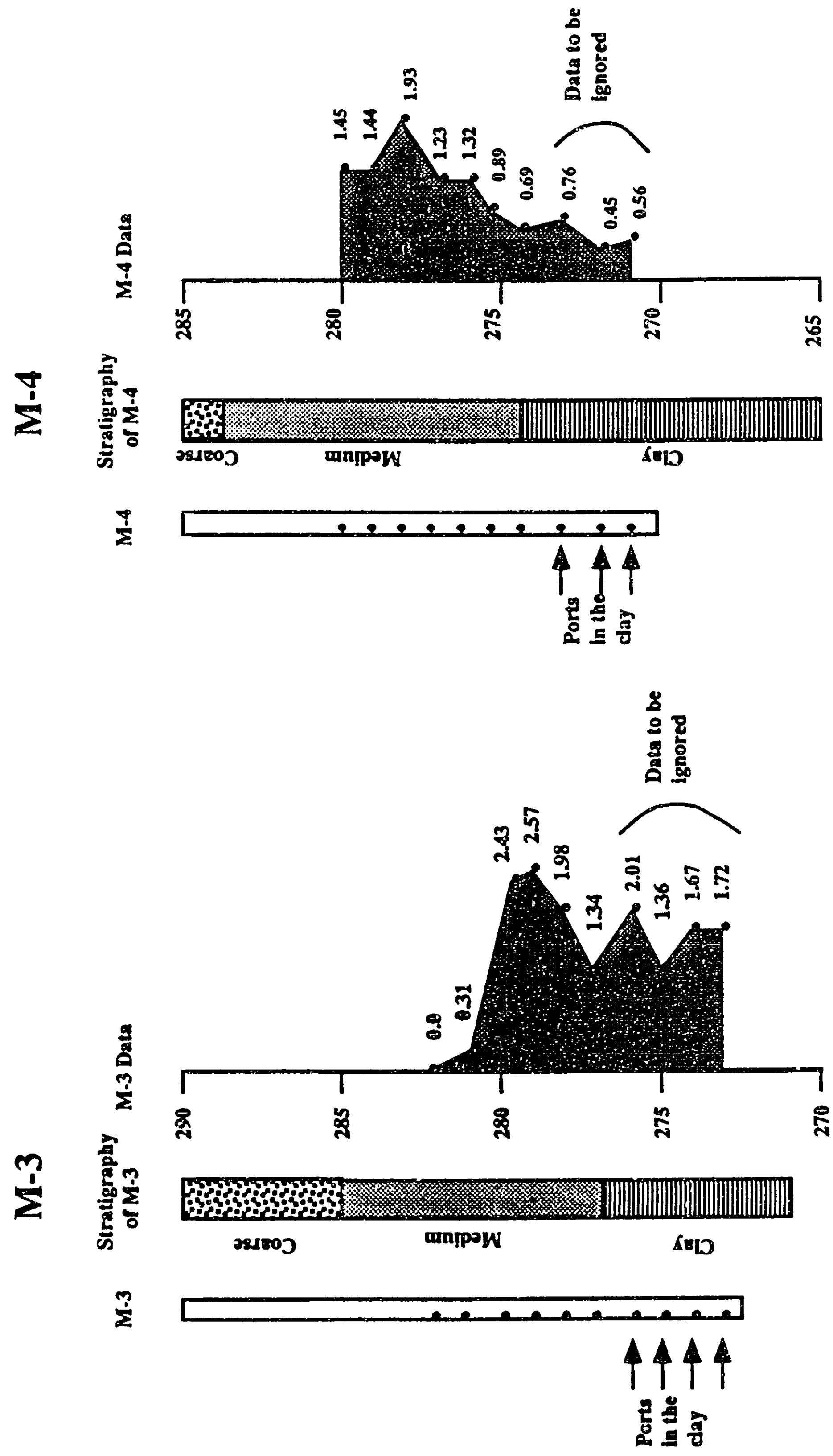


Figure 4-6: Data profiles from MLSs installed into the clay.



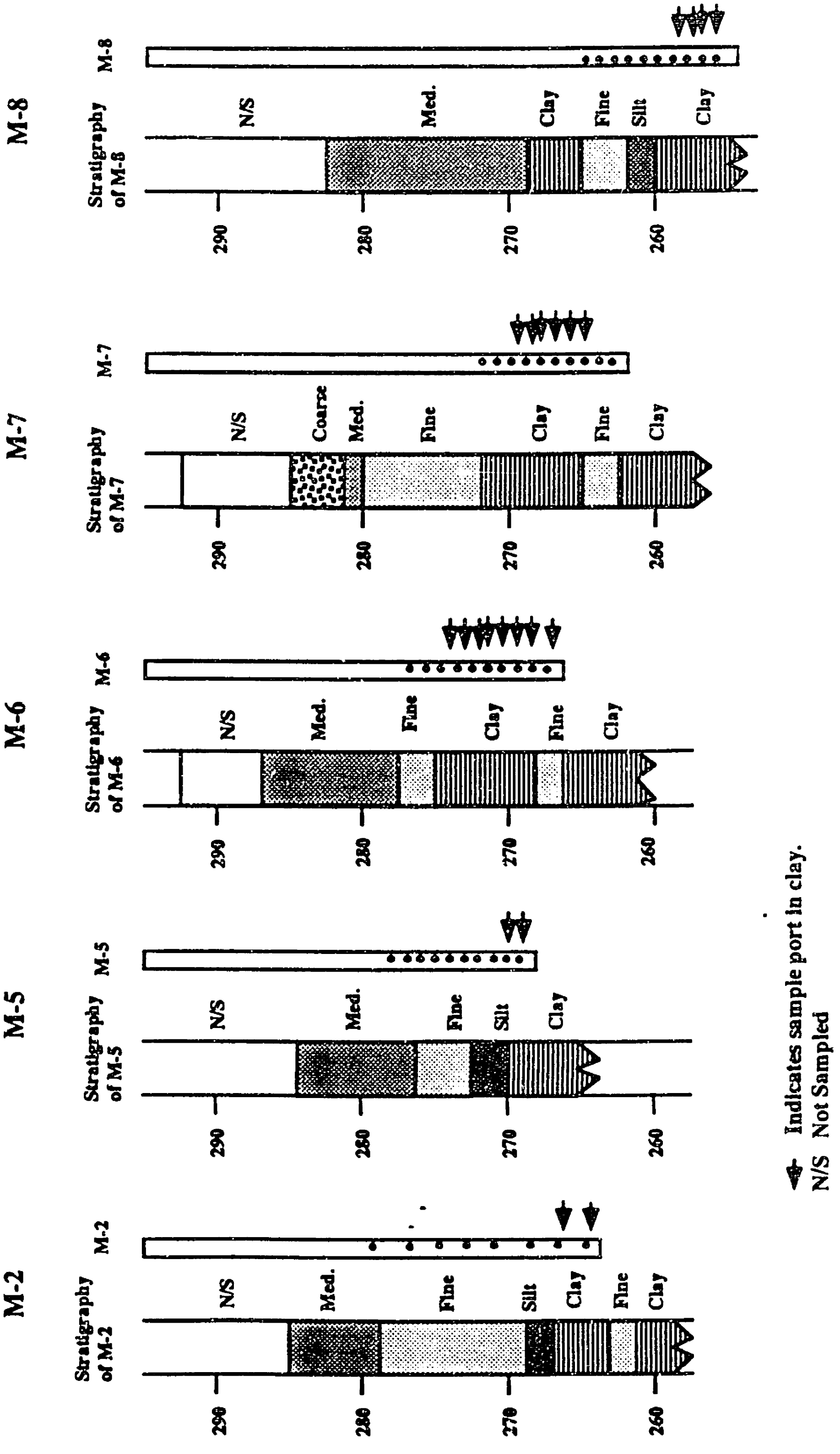
so that the validity of the samples taken above the clay will not be affected. Figure 4-7 shows the other metal MLSs that have been installed into the clay with their approximate soil stratifications. The sampling ports in the clay whose samples will not be used for numerical comparison are shown as marked. The information on the location of soil layers in the aquifer at Site 24 is taken from soil borings performed by Atlantic Environmental. MLSs not displayed in Figures 4-6 and 4-7 are not installed into clay.

4.2 Tracer Tests: Detection of Improper Soil Collapse Around MLSs

If, after installation, aquifer material or backfilled material does not completely collapse around the multi-level sampler, pathways can exist which will create a "short-circuit" between sampling ports. In other words, during sampling when all the ports are being pumped on simultaneously, the two or more ports that are connected in this way will not be taking groundwater samples representative of their particular level. Groundwater will be traveling to the connected ports from areas other than locally via these undesirable pathways. Such a sample would be more like an average of the concentration at the two levels. This situation is also known as "crosstalk".

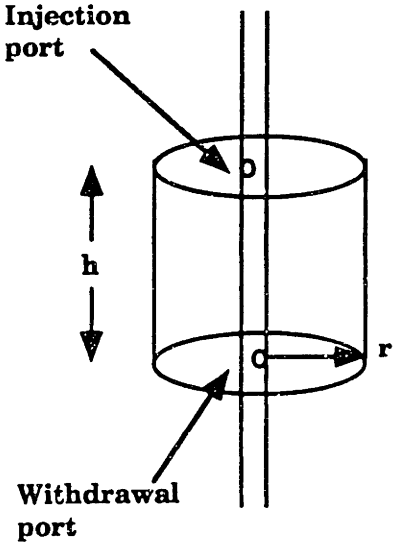
Small scale tracer tests are used to look for signs of short-circuiting along an MLS in situ. The tracer test can take a variety of forms, but the most basic is called a doublet test. In the doublet test, a set volume of tracer is injected into one port while groundwater is pumped out of an adjacent port simultaneously (and at the same rate to conserve mass). In a continuous-injection test water is pumped in behind the tracer until the test is done and in a pulse test, injection stops after the tracer. Short-circuiting is evidenced if the tracer is detected in the withdrawn water in a significantly short amount of time. A period of time that one would expect the tracer to arrive in if no short-circuiting paths exist can be calculated by estimating the time it would take the water to travel through the soil pores in the affected cylinder around the sampler at the rate the test is conducted (continuous

Figure 4-7: The remaining metal MLSs, their soil stratifications and the resulting invalid sampling ports due to the intersection with clay.



injection) (Garabedian, 1987). Figure 4-8 explains this diagrammatically and with a sample calculation. The least certain figure in these calculations is the affected cylinder radius. Once tracer tests are performed, a radius can be calculated using the time the tracer is detected to peak in the withdrawn water. The radius used in Figure 4-8 was taken from the typical radius calculated from tracer tests performed on MLSs at the Cape site (Garabedian, 1987).

A common tracer used is salt, NaCl, which may be detected with a conductivity meter (units of conductance in microseimens) or a chlorine-selective probe. Other tracers include bromide, detectable with a bromide-selective probe, fluorescent tracers, and radioactive tracers.



Travel vol. of tracer = pi x r^2 x h x n

= Time to tracer peak x flow rate

t = (pi x r^2 x h x n) / Q

h = 1 ft = 30.5 cm or 2 ft = 61 cm

Q = 70 ml/min

r = 18 cm

Approx. time to tracer peak for MLS, 1 ft spacing = 2.6 hrs

Approx. time to tracer peak for MLS, 2 ft spacing = 5.2 hrs

Figure 4-8: Sample calculation used in determining the approximate time to peak for a continuous injection doublet tracer test.

4.2.1 Tracer Tests at the Otis Air Force Base, Massachusetts

At the large-scale natural gradient tracer test carried out at the Otis Air Force Base on Cape Cod, small scale doublet tracer tests were conducted on several MLSs to look for crosstalk problems. Three crosstalk tests were carried out using a salt tracer, the results of which are shown in Figure 4-9. In this figure the top and bottom graphs represent tracer tests in which water was recirculated or pumped in behind the tracer. The middle graph, which shows a long gradual return to the background conductivity, did not have water pumped in behind the tracer; the tracer was only a pulse. Theses tests, run at a much higher flow rate than those at Site 24 (about 300 vs 70 ml/min), demonstrate the characteristic behavior of the tracer under both tracer test conditions. These curves also indicate the absence of crosstalk, since there was a significant initial period of time before the tracer arrived (Garabedian, 1987).

4.2.2 Tracer Tests at the Columbus Air Force Base, Mississippi

At the Columbus site in Mississippi where MLSs were exclusively used in a large-scale tracer test, crosstalk and short circuiting were considered as separate problems (TVA, 1988). They defined crosstalk to refer to connections between sampling ports due to improper caving around the MLS, and short-circuiting to refer to local aquifer disturbance caused by MLS installation that affects the ambient groundwater flow. By this definition, Crosstalk interferes with sampling discrete vertical locations in the aquifer, while short-circuiting creates vertical gradients which can lead to vertical mixing and unnatural spreading of the plume.

Two different tracer tests were developed to test for each problem. In the crosstalk test, a bromide tracer slug was pumped into the aquifer just outside of a sampling port. Approximately five minutes after injection the injection port and four ports above and below were pumped slowly. The mass of tracer recovered from each port was kept track of

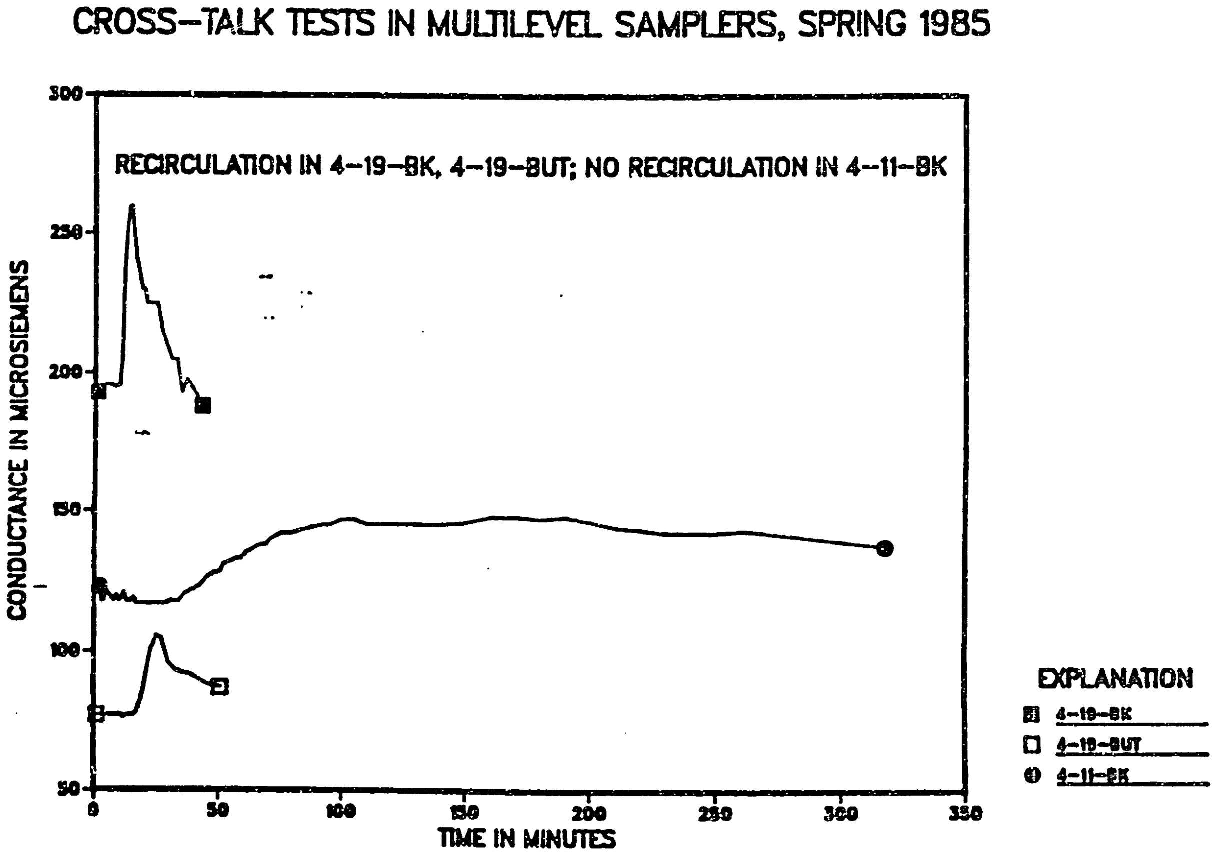


Figure 4-9: Breakthrough curves for doublet tracer tests performed at the Otis Air Force Base, Cape Cod site.

and crosstalk was considered insignificant if less than 5% of the mass was detected from ports other than the injected one. The same protocol was followed in their short-circuit tracer test except a delay of 2 to 22 hours was used between tracer injection and withdrawal. The amount of tracer recovered from each port was used to calculate where the center of mass of the tracer had been transported vertically during the delay period. The distance the center of mass moved from its original injection point was used to calculate a seepage velocity. Short-circuiting was inferred if the seepage velocity was higher than the estimated ambient seepage velocity.

For the augered multi-level sampler similar to the one I built for Site 24, researchers at the Columbus site found evidence of crosstalk in about 30% of the tests conducted (TVA, 1988). Of more concern was the evidence of short-circuiting. Seepage velocity values were in excess of estimated ambient vertical velocity in 62 to 75 percent of the tests. The study found that short-circuit tests on multilevel samplers installed with bentonite packers between sampling ports indicated lower seepage velocities, and had about the same results in crosstalk tests. These results indicate that installation disturbs the area local to the multi-level sampler. It does imply that we do not have a good understanding of the degree to which the disturbed area around the MLS alters the natural properties of the aquifer and in turn, whether this disturbance significantly alters the representativeness of the groundwater sample.

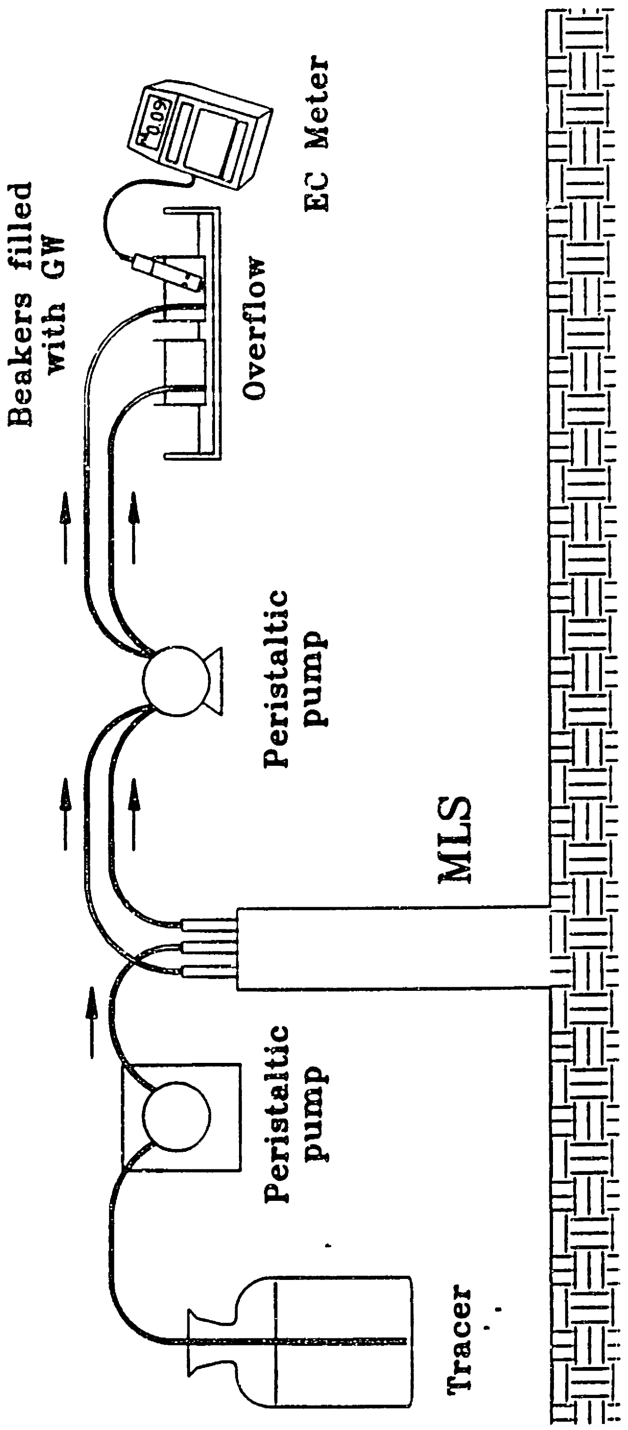
4.2.3 Tracer Tests on Site 24 MLSs

Local tracer tests were performed on a number of the MLSs at Site 24 and on a sample section of MLS in the lab, in an effort to detect crosstalk problems and understand more about patterns of flow around the sampler. Salt was used as the tracer and was detected with a conductivity meter. A simplified diagram of the tracer test set-up used in the lab and at the site is provided in Figure 4-10. Tracer conductivity was mixed to several thousand microseimen (μS) to stand out from the groundwater, having a conductivity of several hundred microseimen.

4.2.3.1 In-lab Tracer Tests

In order to better understand the mechanisms involved in the field tracer tests, continuous-injection doublet tracer tests were carried out in the lab on a a typical section of MLS having 2 ports, 1 foot apart. The MLS section was placed in a large container of homogeneous medium-grained sand. Tracer tests were run at various flow rates to completion, achieving breakthrough (when the tracer initially reaches the withdrawal port), passing a peak (in conductivity), and finally trailing off. Results of four lab doublet tracer

Figure 4-10: A simplified diagram of the tracer test set-up at Site 24.



tests are shown in Figure 4-11. In these tests the flow rate was varied in hopes of establishing a proportionality between rate of flow and time to peak. The tracer was injected in both the top and bottom ports to see if gravity had an affect on the tracer's transport. The graphs show it takes from 2 to 3 hours to perform a complete tracer test. The graphs are inconclusive, however in determining whether gravity has an effect on the transport of the tracer.

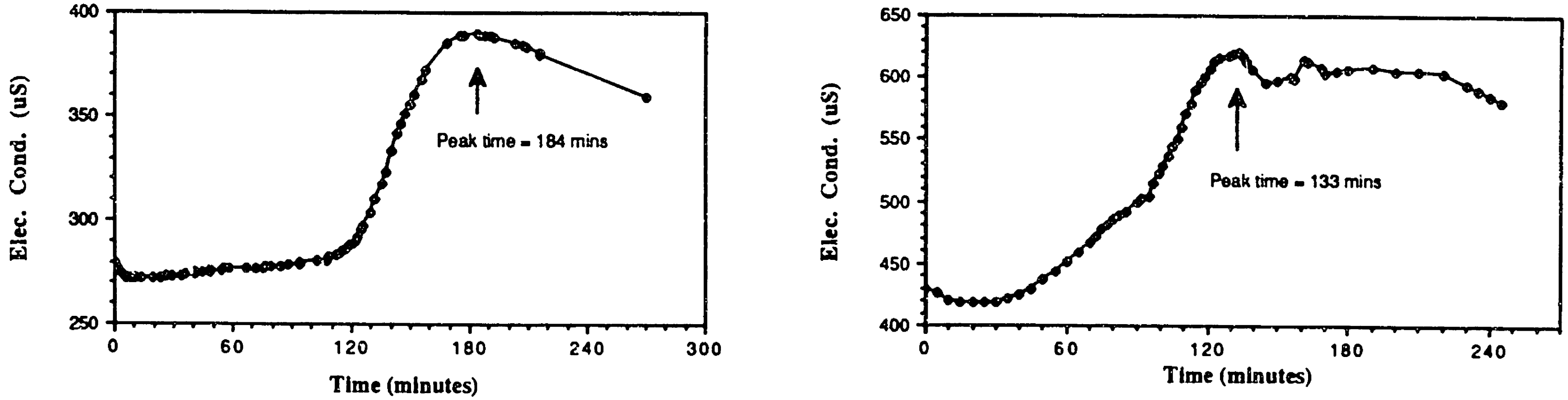
We found that complete tracer tests run in situ can take even longer than three hours. It is not necessary, however, to run a complete tracer test in order to demonstrate the absence of crosstalk. Because tracer tests are run at flow rates over twice that used for taking samples, if no evidence of the tracer (breakthrough) is seen in well over the time it takes to sample an MLS (an hour leaves a good margin), then it can safely be assumed that no crosstalk channels exist that would interfere with sampling.

4.2.3.2 Field Tracer Tests

Pulse tracer tests were initially used at Site 24. Tracer tests were carried out after sampling so that the salt tracer did not interfere with the groundwater chemistry. Flow rates of between 40 and 90 ml/min were used because they were the fastest rates that allowed continuous withdrawal of groundwater from the aquifer. Faster flow rates did not allow a continuous flow of water due to the limited specific yield of the aquifer.

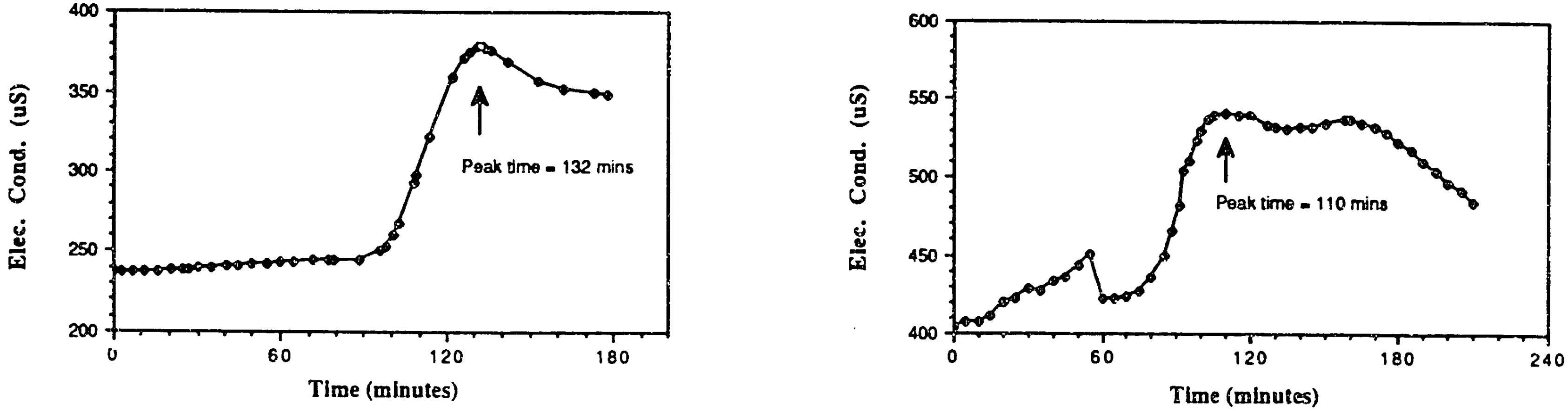
The first tracer tests carried out involved injection of a slug of tracer (about 3000 µS) into one port and then withdrawal from the two vertically adjacent ports above and below (5-port test) (see Figures 4-12, 4-13, and 4-14 part a). A second type of tracer test used three ports (triplet test), with the center one for injection of the tracer and recirculated water (continuous injection) and the outer adjacent two for withdrawal (see Figures 4-14, parts b and c). The triplet test and the 5-port test examined two port connections at once. The last type of tracer tests run were identical to the doublet tests performed at the Cape site and demonstrated in the lab, (described above). These tests pump a tracer followed by recirculated water into one port, and withdraw from one adjacent port (see Figure 4-15).

Figure 4-11: In-lab continuous-injection doublet tracer tests results.



a) In-lab Doublet Tracer Test, Q = 70 ml/min, In high/ out low

c) In-Lab Doublet Tracer Test, Q = 88 ml/min, In low/ out high



b) In-lab Doublet Tracer Test, Q = 77 ml/min, In high/ out low

d) In-lab Doublet Tracer Test, Q=114 ml/min, In low/ out high

Of the ten graphs shown in the four figures mentioned above, only Figure 4-13, part a), shows evidence of crosstalk. In this test, conductivity of the water injected from port 12, withdrawn from port 14, rose immediately, jumping 300 µS in less than 20 minutes. This indicates a high conductivity connection between ports 12 and 14 of P-1 (The presence of crosstalk here will not affect the data analysis, since data from the plastic-tubed MLSs will not be included in the final analysis due to sorption). The tracer breakthrough depicted in Figure 4-12, part b), is probably not indicative of a crosstalk problem that will interfere with local sampling because the tracer took almost an hour to travel from injection to withdrawal, and did so at 70 ml/min, a rate more than twice that used in sampling. The small variations in the conductivity of the withdrawn groundwater displayed in some of these graphs is most likely due to slight variations in water chemistry, or measurement variation associated with the conductivity meter. The results of the tracer tests, one evidence of crosstalk out of 17 port connections tested, indicate that in general, the soils around the MLSs installed in Site 24 have caved properly, and groundwater samples taken are likely representative of the area local to each sampling port.

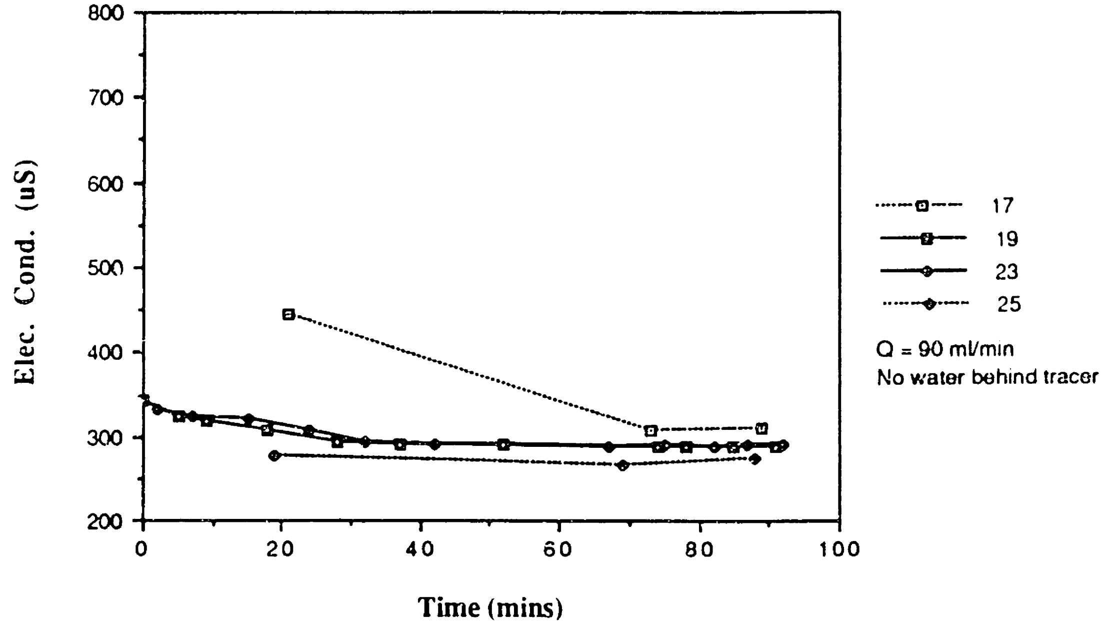
4.3 Interpretation of Data

The data taken at Site 24 on the concentration of naphthalene in the aquifer come from groundwater taken from multi-level samplers and monitoring wells (MWs) and soil taken from drilling installation of the same monitoring wells and piezometers. With the ultimate goal of delineating the plume in the vertical, longitudinal, and transverse directions, it would be optimal if all these three types of data could be used together towards this end. This section will attempt to relate these three different data types (MLS, MWs and soil), and then select the way of best using these data to obtain the most accurate image of the naphthalene plume in three-dimensions.

Monitoring well data were taken at Site 24 by employees of CAA by peristaltic pump

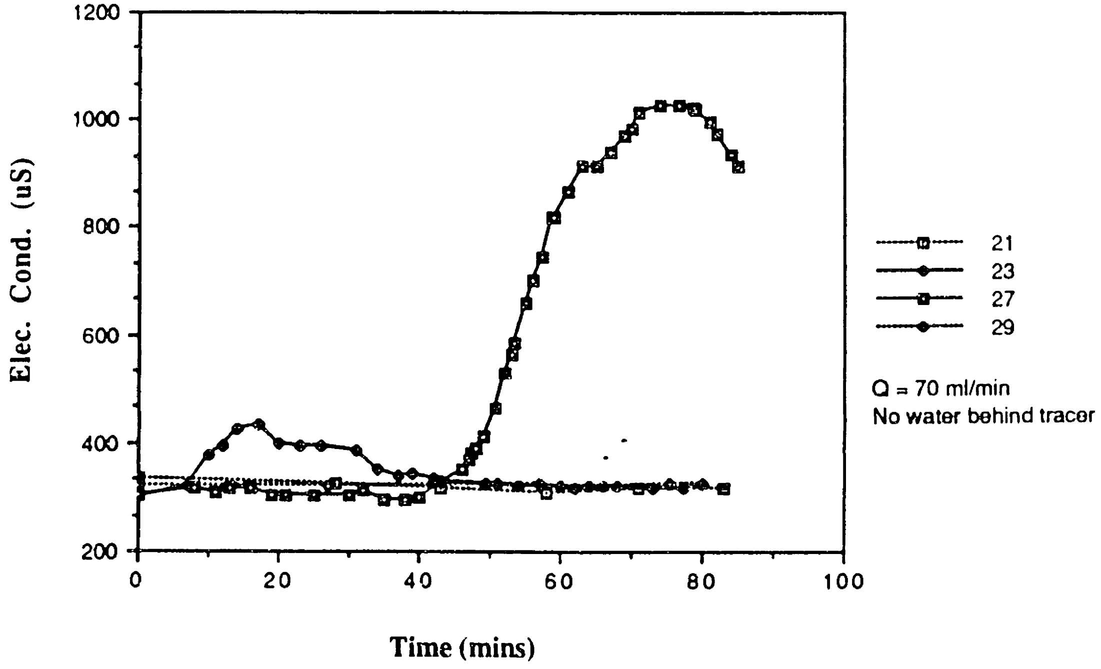
Figure 4-12: Tracer tests performed on MLS M-2.

a) Tracer Test at M-2, in at Port 21



Q = 90 ml/min
No water behind tracer

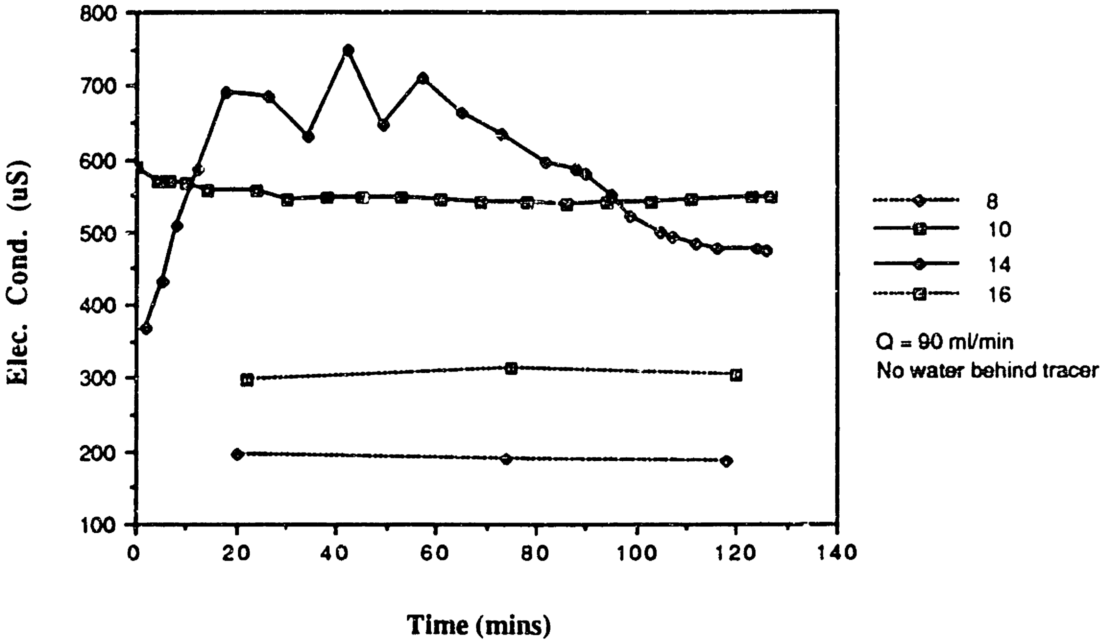
b) Tracer Test at M-2, in at Port 25



Q = 70 ml/min
No water behind tracer

Figure 4-13: Tracer tests performed on MLS P-1 and M-1.

a) Tracer Test at P-1, in at Port 12



b) Tracer Test at M-1, in at Port 16

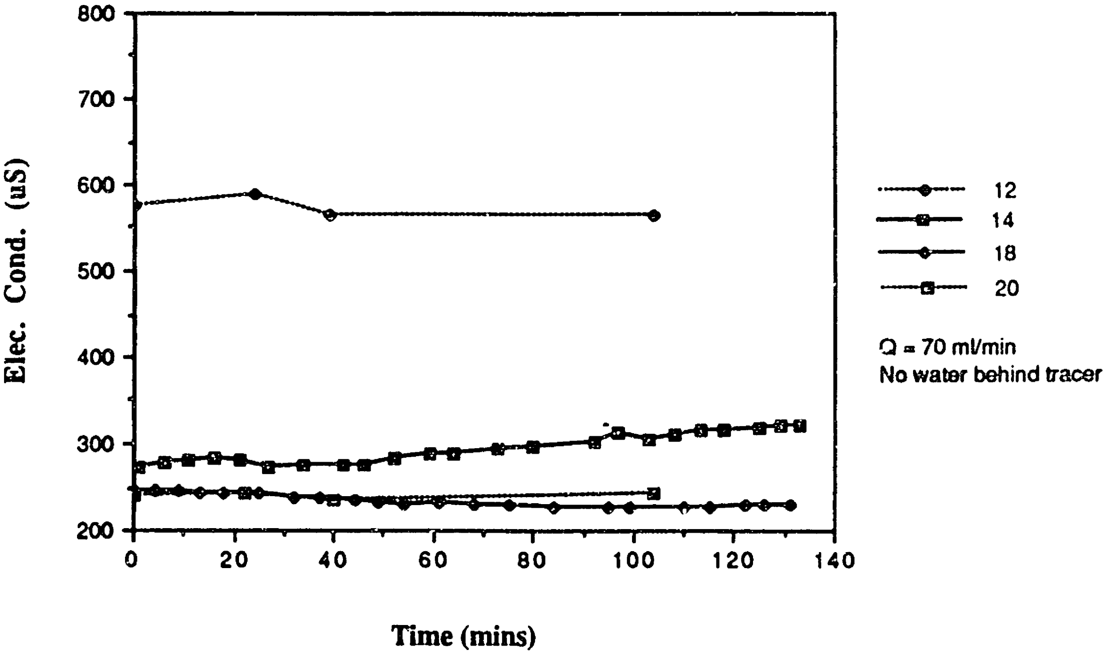
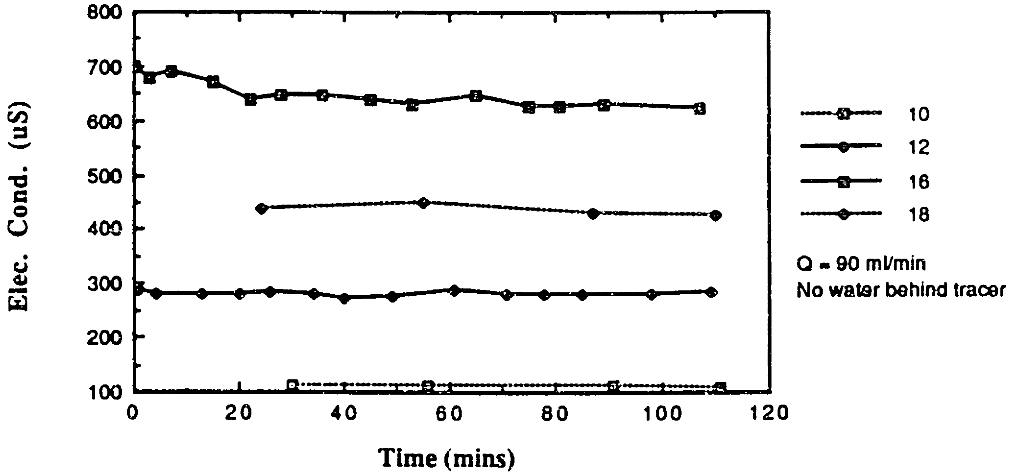
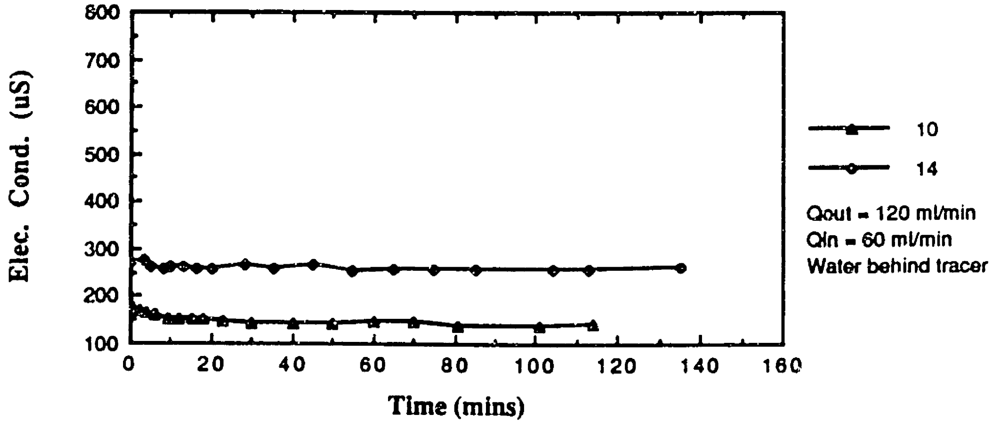


Figure 4-14: Tracer tests performed on MLS P-2.

a) Tracer Test at P-2, in at Port 14



b) Tracer Test at P-2 in at Port 12



c) Tracer test at P-2, in at Port 18

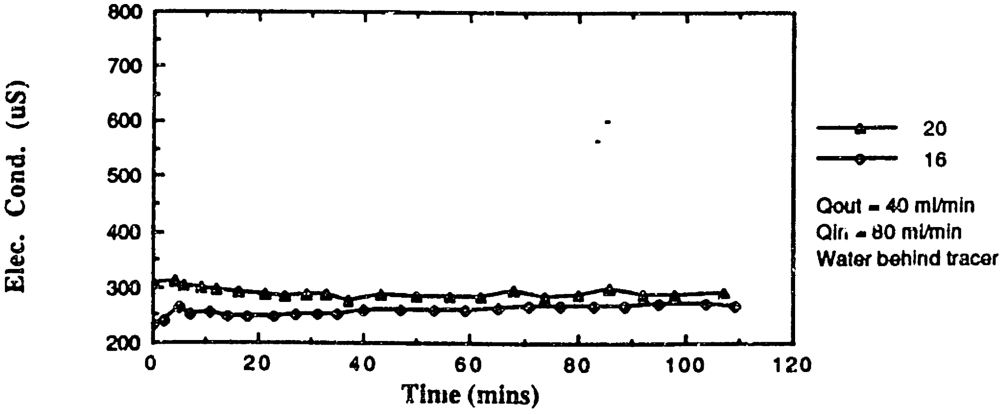
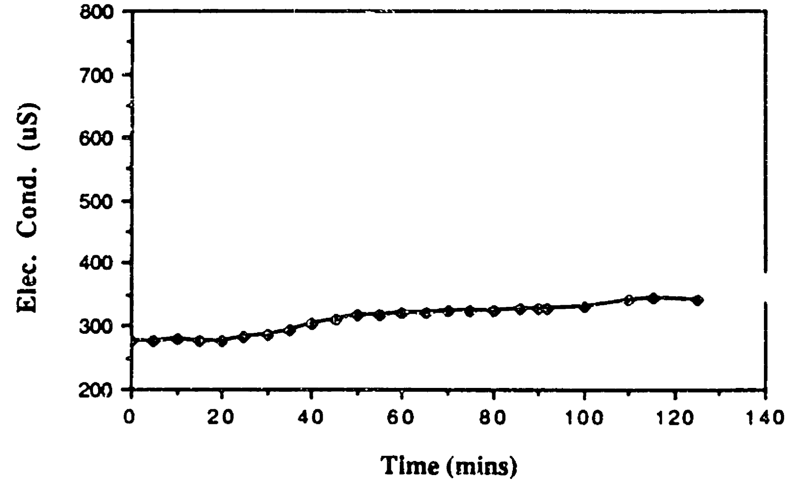


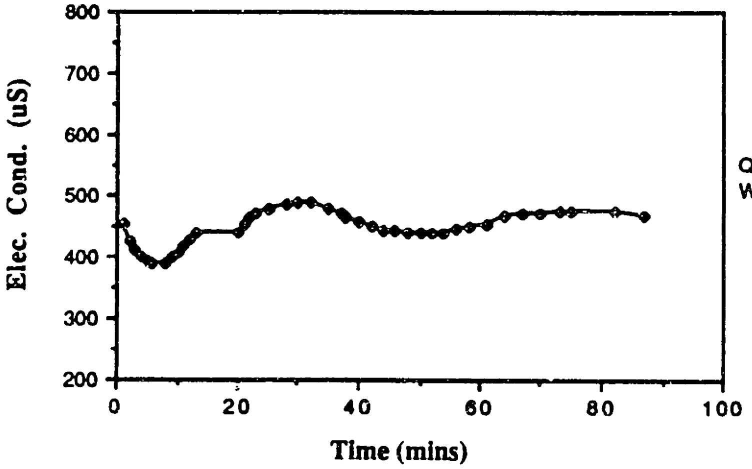
Figure 4-15: Tracer tests performed on MLS M-3 and M-4.

a) Tracer Test at M-3, in at Port 2, out at Port 3



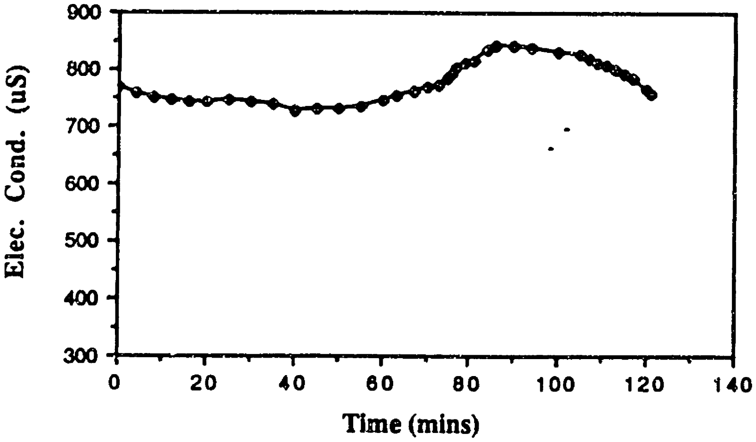
Q = 70 ml/min
Water behind tracer

b) Tracer Test at M-3, in at Port 5, out at Port 6



Q = 70 ml/min
Water behind tracer

c) Tracer Test at M-4, in at Port 1, out at Port 2



Q = 54 ml/min
Water behind tracer

and by bailing. In taking a pumped sample, they placed the peristaltic pump before the sample vial (unlike the special sampling set up I used to sample the MLSs, see Figure 3-13). Soil samples were taken by CAA using a split spoon device, a rigid metal tube that is driven into the aquifer and preserves a 2.5 foot vertical section of soil. Soil data are given in terms of mg of naphthalene per kg of wet soil (same as ppm) and solute data are in terms of mg of naphthalene per liter of groundwater (same as ppm). All the data presented in this chapter are analyzed by GC/FID unless specified otherwise.

As was mentioned in the introduction of this thesis, the main reason for taking data on the concentration of toxic contaminants in the groundwater in the real world is for the assessment of risk at a site, or to describe accurately the places of the highest levels of contamination if they must be remediated or removed. To assess risk, it is important to know how much of the toxins are where so that the future of the site can be determined, i.e. the site can be left alone, should be fenced off at these locations, or should be remediated. If the site must be remediated, it will save money to know where the different levels of contamination are so that soil that is not contaminated is not treated.

4.3.1 Assessment of MLS Data as 2-Dimensional Data

Figure 4-16 shows the two-dimensional data derived from the first round of well stations and MLSs installed at Site 24. The data portrayed on this map were all analyzed by GC/FID by CAA and is the highest value of all the data taken from monitoring wells (by pumping, bailing, etc.) in 11/88 and 6/89 and from MLSs in 6/89. After these few rounds of sampling, the data indicate that the plume is narrow near the source (since stations 6 and 8 are very low) and that the centerline near the source curves up to go through station 5 between stations 1 and 7. The high observations between 1-2 ppm at M-1 and P-2 could indicate that the centerline is fairly horizontal from sta. 7 on east. The lack of data along transects C and beyond allows no conclusions to be drawn on the plume boundaries at this distance away from the source.

After the second round of MLSs and monitoring wells were installed and sampled, a more detailed 2-D image of the plume surfaced. A number of well stations were specifically installed by CAA at the extremes of the transects in order to locate the (north and south) edges of the plume. With this more advanced round of data, shown in Figure 4-17, the edges of the plume can be guessed with an accuracy of a couple hundred feet. The centerline of the plume seems to dogleg more severely as the new data from M-3 shows, and in general the centerline, though not clearly defined, seems to follow a crooked path from the source to the seeps.

With the last round of four MLSs installed and sampled, a very good image of the plume is developed, shown in Figure 4-18. Because at this point the wells and samplers are spread out no more than 200 feet along a transect, the centerline and the top and bottom edges of the plume can almost be resolved. The plume is thin and its centerline meanders a bit, but goes directly from the source to the seeps without straying.

Given the broad concentration ranges spanning 0.5 and 1 ppms specified in Figures 4-16, 4-17 and 4-18, the MLS data is as useful as the monitoring well data. If more accuracy is desired, data points from specific elevations along the MLS must be compared with data from specific elevations in the well station.

4.3.2 Vertical Comparison of MLS and MW Data

Four MLSs are installed at Site 24 within 40 feet of a well station. M-3 is about 15 feet from Sta. 5, M-5 is about 18 feet from Sta. 19, M-2 is about 10 feet away from Sta. 15 and M-8 is about 40 feet from Sta. 13. Diagrams comparing the data from these samplers adjacent to well stations are shown in Figures 4-19 and 4-20. These figures do not include MLS data from sampling ports installed into the clay zone for reasons described previously (see Section 4.1.6, "Consideration of MLSs Installed Into the Clay", on page 75). Elevations in these figures shown along the well station data are in feet, and naphthalene

Figure 4-16: Highest naphthalene concentrations taken from MLSs and MWs after installation Round 1.

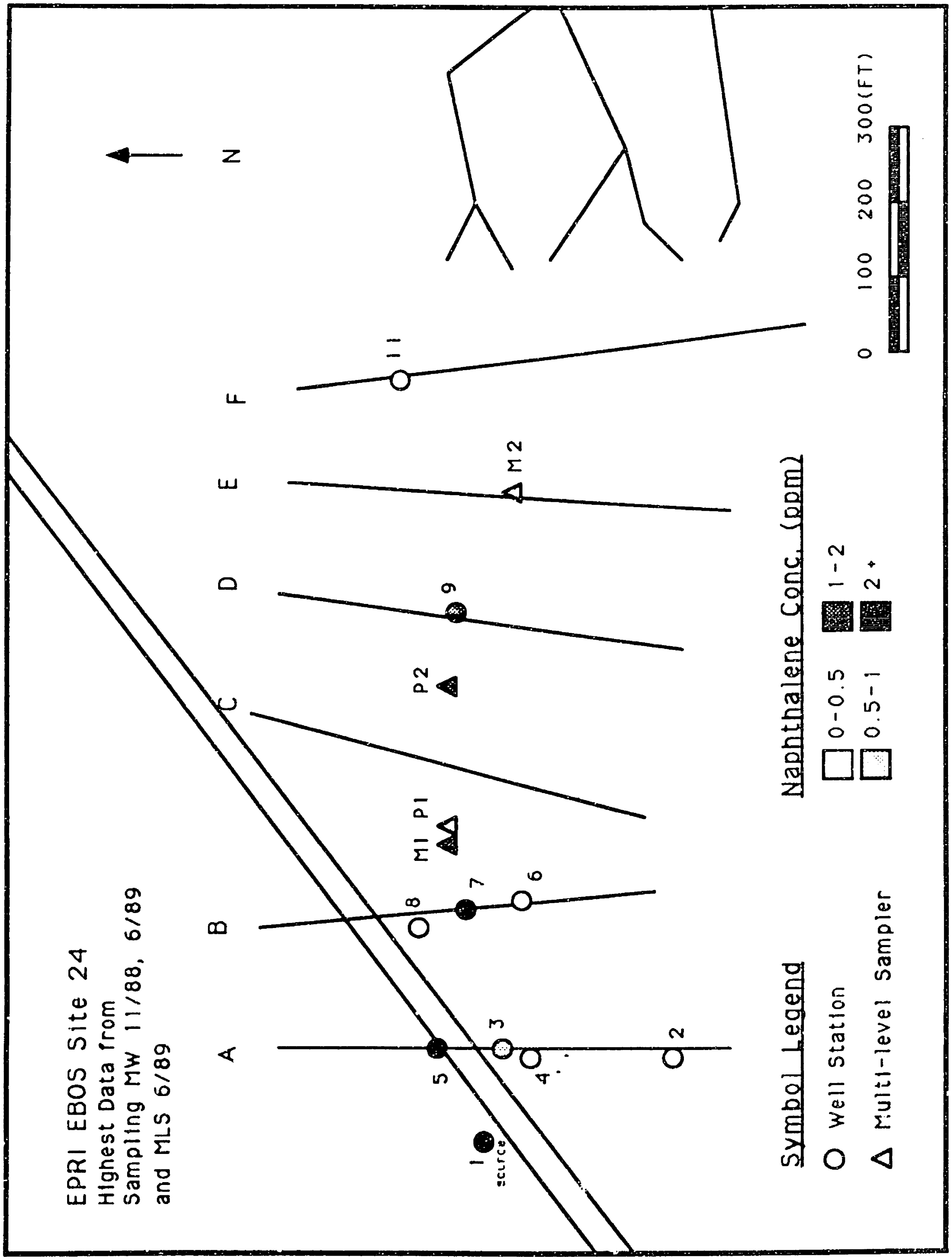


Figure 4-17: Highest naphthalene concentrations taken from MLSs and MWs after installation Round 2.

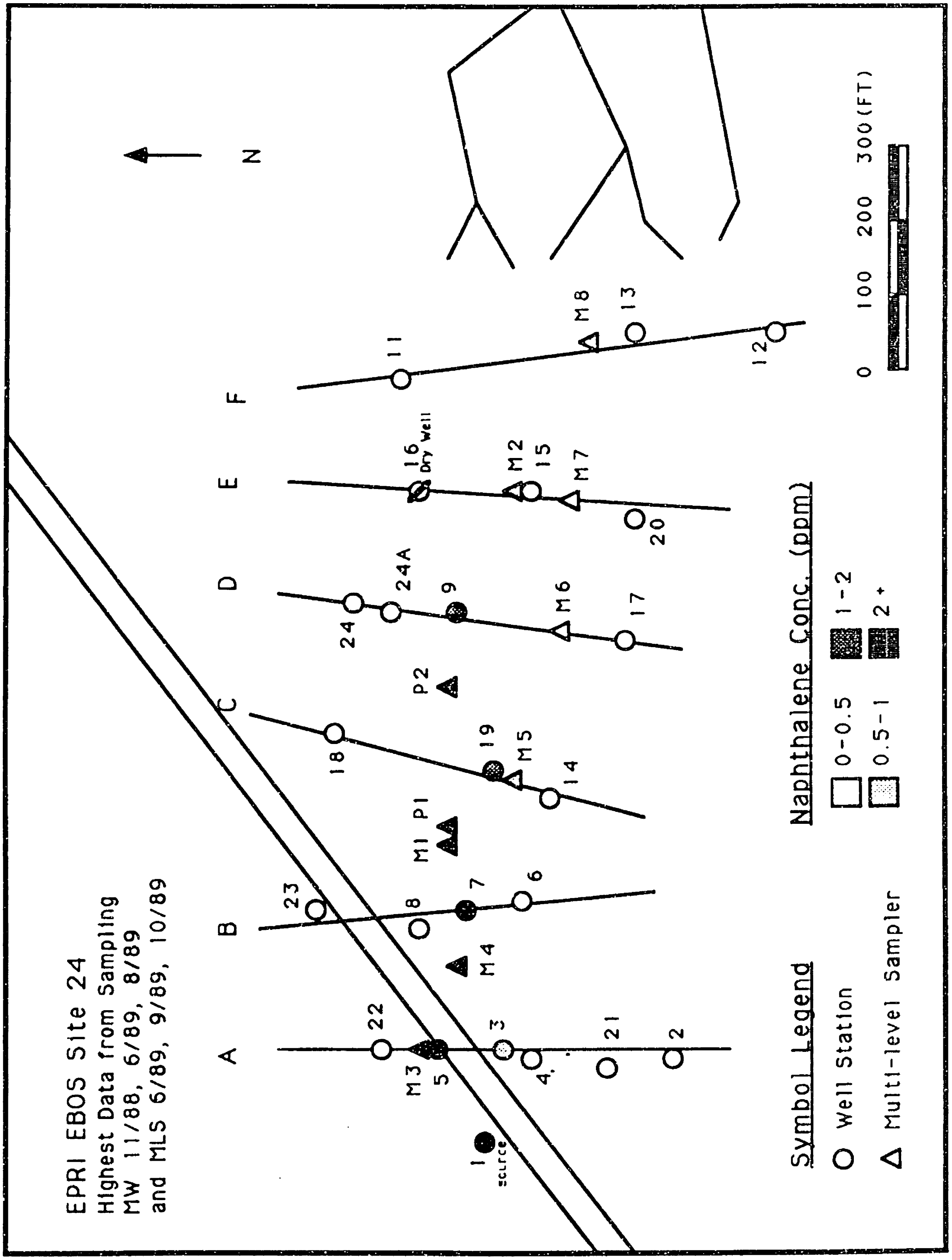
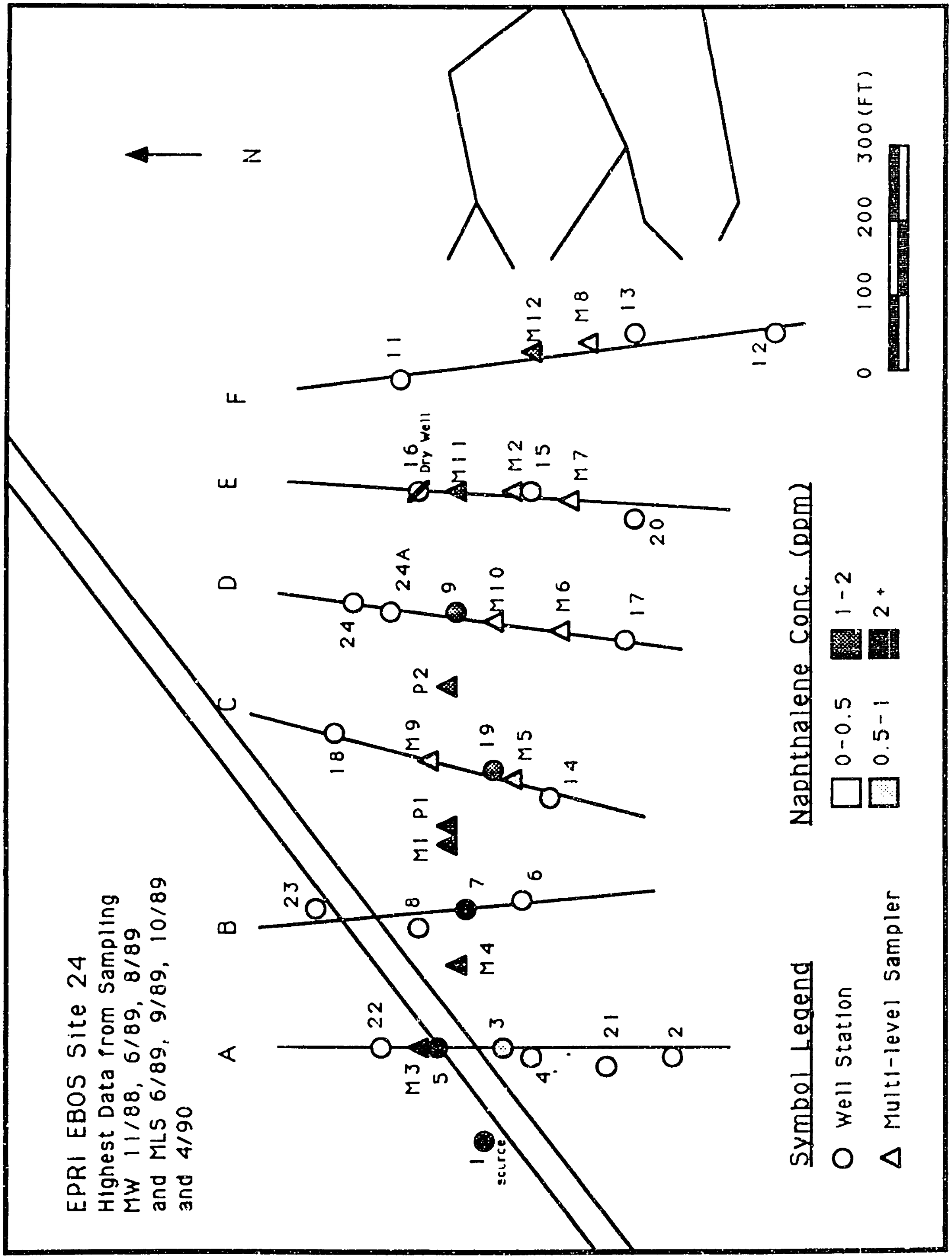


Figure 4-18: Highest naphthalene concentrations taken from MLSs and MWs after installation Round 3.



concentrations shown are in ppm. Soils data are also included in these figures and will be discussed in the next section, Vertical Comparison of Solute and Soil Data.

What we expect to see is that the monitoring well data is an average of the MLS concentration points that cover the same elevations. This should be more apparent for the well stations that are closest to MLSs, i.e. those shown in Figure 4-19. The differences in data from MLSs and well stations that are further apart may be more likely due to sampling different parts of the plume.

The comparison of the farthest samples, in part b) of Figure 4-20, is the best example of the expected relationship between the MLS and monitoring well data. The MLS M-8 data peaks at 0.52 ppm over several feet of elevation while nearby at Sta. 13, the bottom well screen at the same elevation gives a lower, averaged value of 0.32 ppm. The top well screen at Sta. 13 gave a concentration of 0.05 ppm, an average of the two MLS data points at that elevation, 0.0 and 0.11 ppm. The top well at Sta. 15, in part a) Figure 4-19, is another example of this averaging behavior. This well has a concentration of 0.01 ppm while the MLS data at that level jumps from below that value at 0.006 ppm to above that value at 0.019 ppm. The last example of monitoring well averaging of MLS data in these figures is the lower well of Sta. 5 in part b) Figure 4-19. The monitoring well concentration of 1.72 ppm is close to an average of the MLS data points at that elevation, 1.34, 1.98 and 2.57 ppm.

About half of the data on these figures does not follow the expected trend mentioned above. Generally speaking, the MLS data are less than corresponding monitoring well data. This could be explained by the unpredictable variations associated with the imprecision inherent in sampling and analysis. For MLS M-5 at which the concentrations are all lower than those at the adjacent monitoring well, the variation in the plume concentrations over the 18 foot distance between them is probably responsible for the unexpected difference.

Figure 4-19: Comparisons of soil and groundwater data from adjacent MLSs and Well Stations.

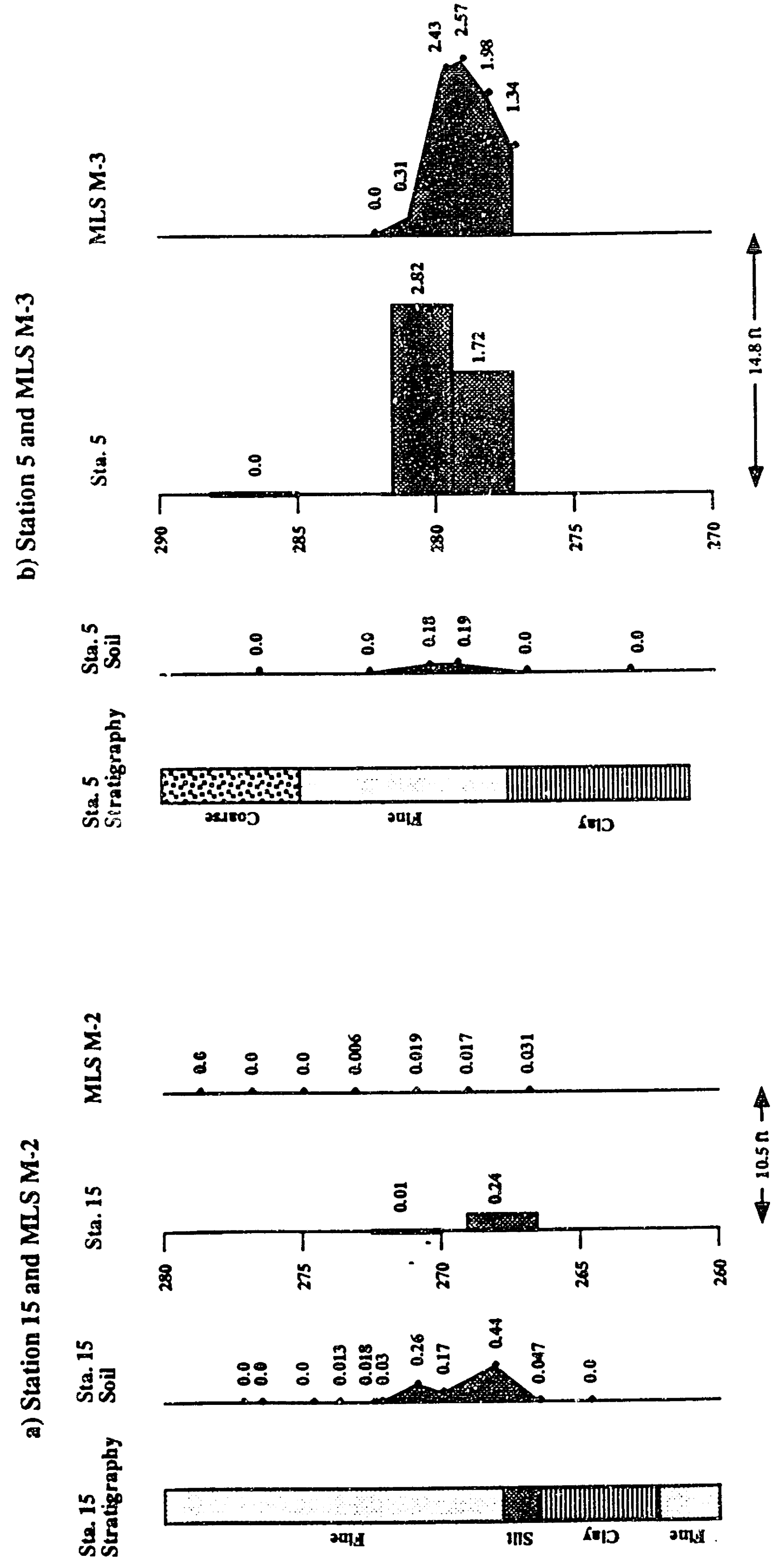
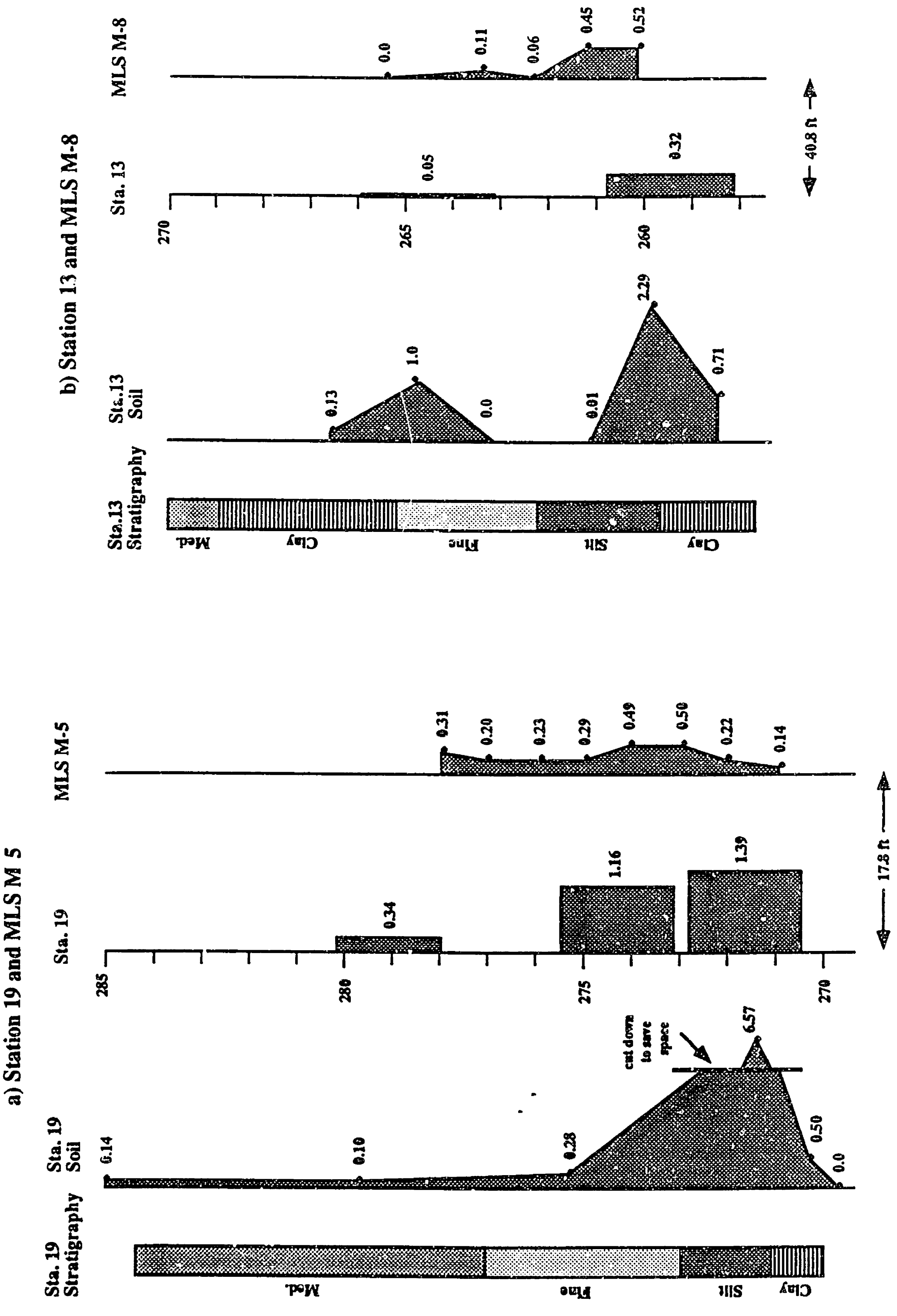


Figure 4-20: More comparisons of soil and groundwater data from adjacent MLSs and Well Stations.



4.3.3 Vertical Comparison of Solute and Soil Data

Out of the four comparisons shown in Figures 4-19 and 4-20, only in part b) of the first figure is the peak soil concentration lower than the solute concentration. In general it seems that the concentration of naphthalene in the soil is higher than in the groundwater. For the most part, however, the soil data is in no predictable way related to the solute data. One way the soil data is useful is that naphthalene is detected in the soil where it is also detected in the groundwater, allowing the soil data to show the extent of contamination, if not the degree. Because of this feature, the soil data was used at this site during well installation to tell where the screens should be located in a borehole. As the borehole was drilled, soil samples were taken at different depths. The soil was quickly analyzed and where contamination was detected, well screens were placed. Although such a technique involves drilling the hole twice (because while the soil is being analyzed, the borehole usually caves in), more data is gained and the wells are installed where they may be more informative.

In the third round of MLS installation soil data were used for siting. Figure 4-21 shows the data from M-10 and M-11, MLSs which were installed particularly close to where soil samples had been taken. This figure supports the point made from the previous figure, that the soil data do not correlate well with the solute data, but are useful for approximating the vertical extent of contamination. The soil data was very effective at predicting where naphthalene would be detected strongly in the groundwater.

4.3.4 The Use of Transects of MW and MLS Data to Create a 3-dimensional Picture of the Plume

The easiest way to imagine a three-dimensional object is to cut it up into slices, look at the slices consecutively, and put together a three-dimensional picture in your head. To look at a whole plume in three dimensions, concentration data may be examined along

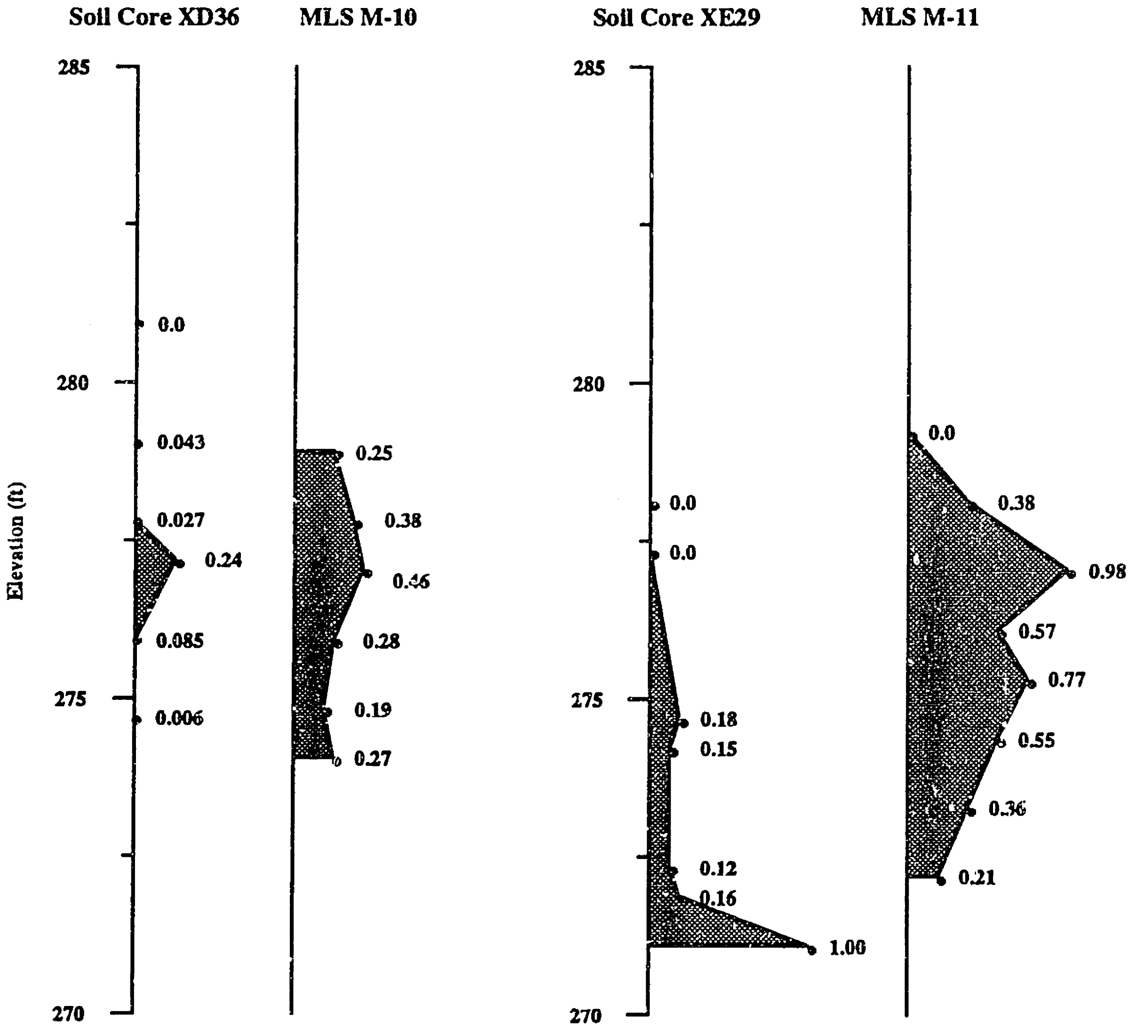


Figure 4-21: A comparison of adjacent soil and MLS data, separated by about 5 to 10 feet.

transects that slice the plume a number of times. Figures 4-22, 4-23, 4-24, 4-25, 4-26, 4-27 and 4-28 show sections through the plume at Site 24 at transects A, B, C, D, E, F and through the centerline, respectively (refer to Figure 4-18 for the plan of Site 24 showing the location of the transects and the "centerline"). The monitoring well data shown is an average of all data taken by CAA by pumping or stainless steel bailer and analyzed by GC/FID. The MLS data from MLSs that were sampled more than once are also an average. All concentrations are for naphthalene in ppm, all elevations in feet. Data from MLS ports in clay are omitted.

Figure 4-22: A cross-section of the naphthalene plume through transect A.

Figure 4-23: A cross-section of the naphthalene plume through transect B.

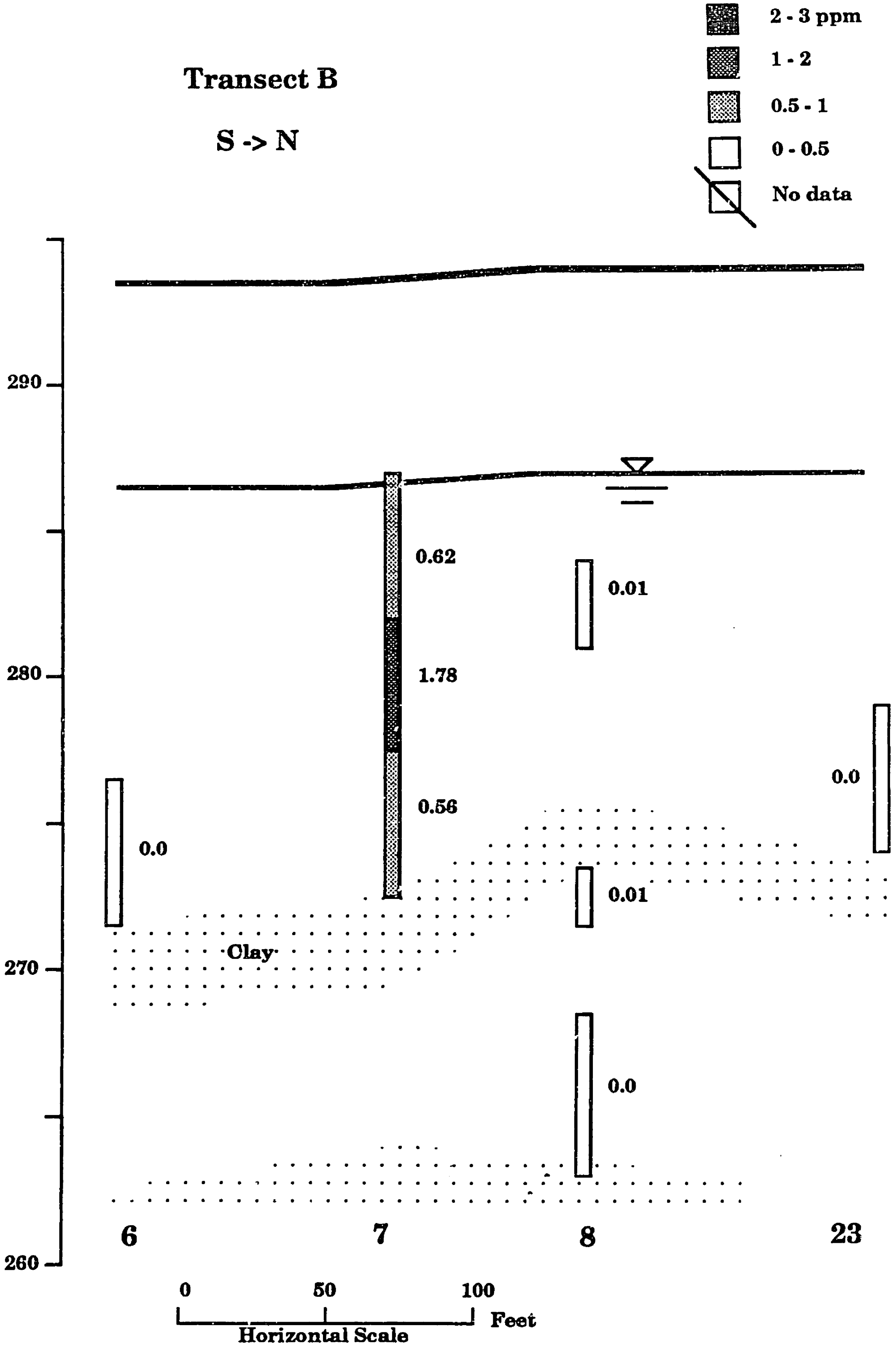


Figure 4-24: A cross-section of the naphthalene plume through transect C.

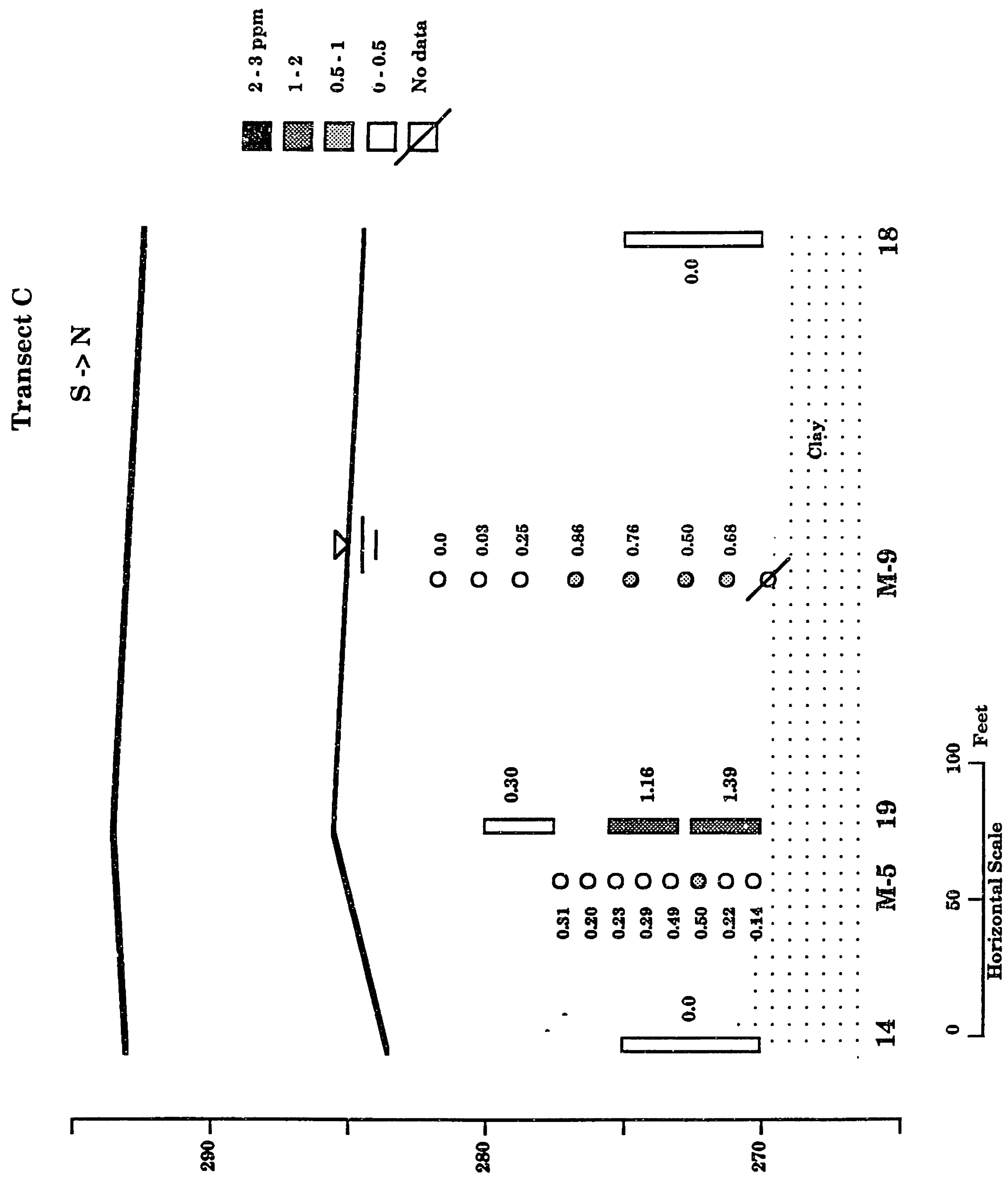


Figure 4-25: A cross-section of the naphthalene plume through transect D.

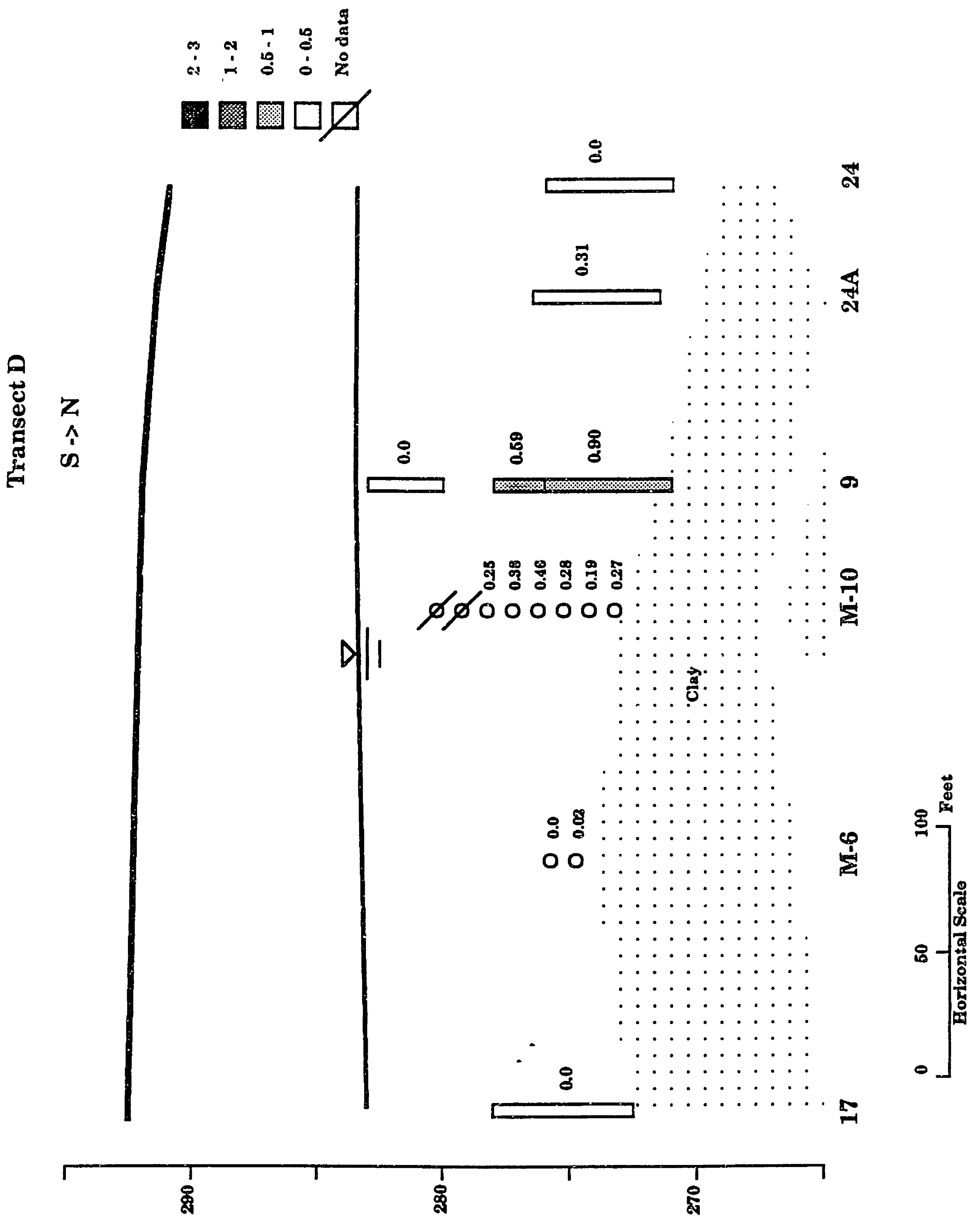


Figure 4-26: A cross-section of the naphthalene plume through transect E.

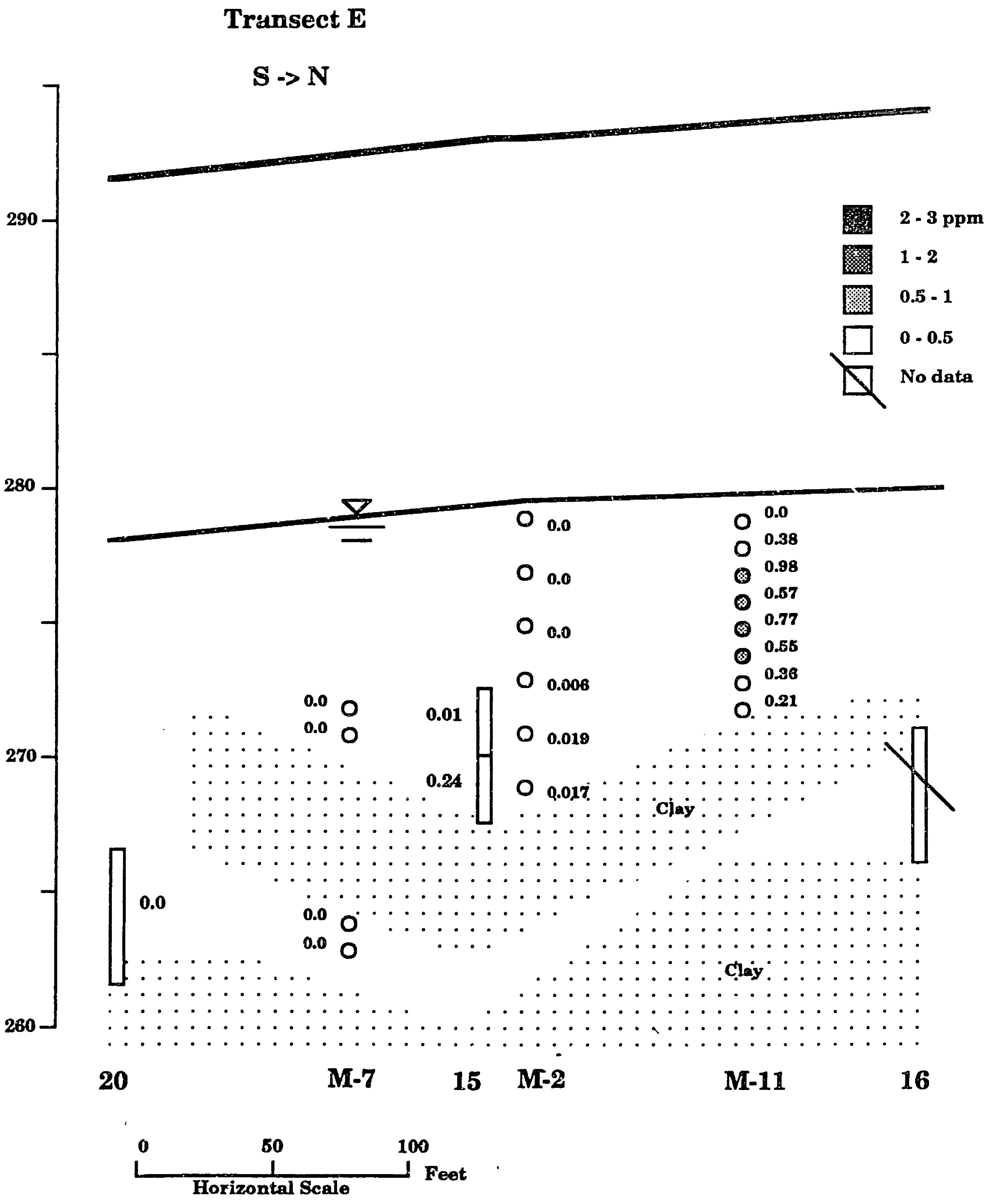


Figure 4-27: A cross-section of the naphthalene plume through transect F.

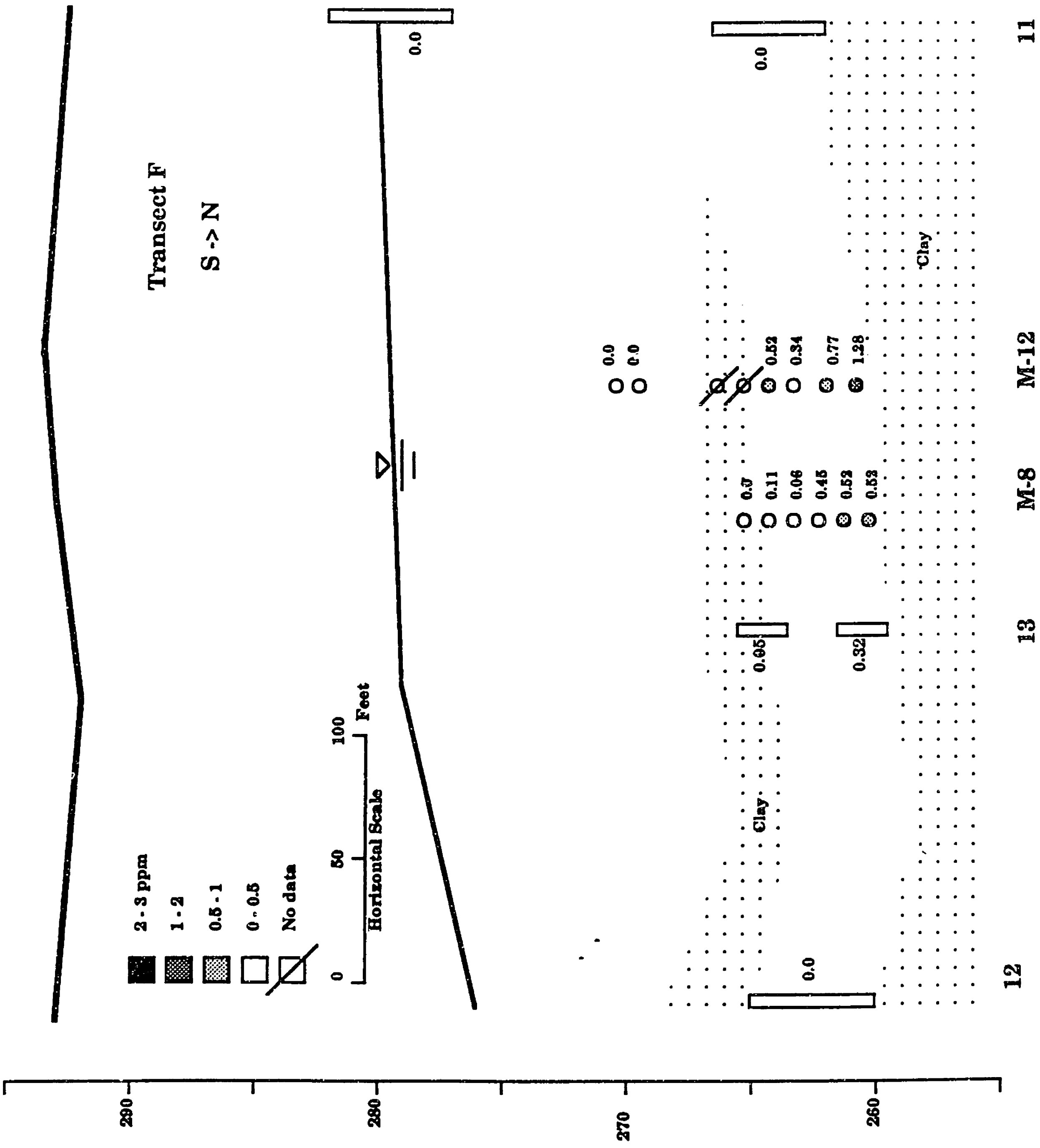


Figure 4-28: A cross-section of the naphthalene plume through the "centerline".

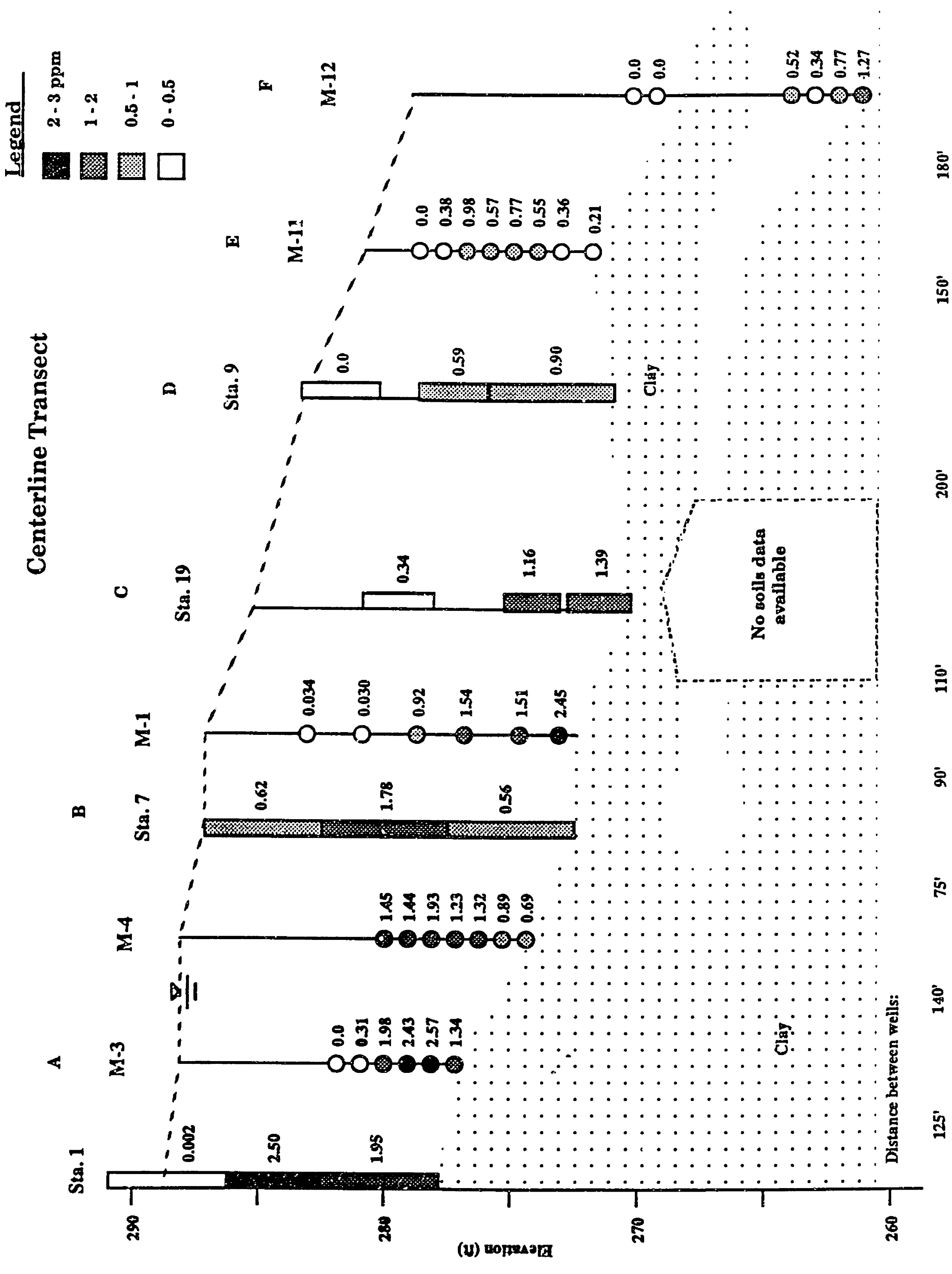


Figure 4-22 shows that the peak concentrations of naphthalene coming off the source are in the range 2.5- 3.0 ppm. On transect A the plume is very shallow (5-9 feet deep), since the aquifer is shallow (about 15 feet deep). A clay lens breaks away from the thick clay layer beginning below Stations 2, 21, 4, and 3.

Along transect B, shown in Figure 4-23, the clay lens and bottom clay layer can be seen. Station 8, with wells below the clay lens, shows that a small amount of contamination (0.01 ppm) has made its way below the lens. The bulk of the contamination (up to 1.78 ppm)is detected above the lens. The contamination plume spans the aquifer above the lens, about 15 feet, but is narrow, delineated by the non-detects in stations 6 and 8.

On transect C significant amounts of naphthalene (up to 1.39 ppm) were detected above the clay lens, as shown in Figure 4-24. With concentrations as high as 0.5 ppm over 100 feet apart, the plume seems to have spread in the horizontal direction. No information is known about soil types or naphthalene concentration below the first clay layer reached.

Figure 4-25 displays the clay layers and naphthalene concentrations detected along transect D. A very narrow break in the clay (2 feet high- see Figure 4-28) was detected below Station 9, but sampling wells were not installed to sample such a depth. Naphthalene was detected at high levels above the clay (up to 0.90 ppm).

The picture of transect E shown in Figure 4-26 displays again the presence of the clay lens. Contamination is detected above the lens (up to 0.98 ppm) but samplers installed below the lens were too far away from the plume to detect contamination (stations 20 and M-7) or had problems (station 16).

The wells and MLSs installed along transect F, shown in Figure 4-27, all intersect the plume below the clay lens, which is the widest along this transect (4 -5 feet). The two top ports of M-12 indicate that no contamination is present above the clay lens. Naphthalene as high as 1.28 ppm was detected by M-12 below the lens.

Finally, 4-28 shows a diagram of the concentrations of naphthalene in the vertical from the well stations and MLSs with the highest concentrations, the hypothetical centerline. In this figure one can see that the height of the plume changes quite a bit, from 5 feet high at transect A to 15 feet at transect B. The naphthalene plume appears to stay as low in the aquifer as it is able, possibly due to the significant vertical head gradients in the aquifer. From the data it is ambiguous where the contamination was able to seep through the clay lens. Because so much contamination was detected below the clay lens, it is not likely that holes punched through the lens during well installation are responsible. The location of the clay layer is based on a number of soil cores taken along each transects. The location of the clay layers shown in the plume cross-sections are extrapolated from these measurements. It is possible that breaks in the clay exist between the transects or below transect C where soil cores were not taken.

Multi-level samplers were more useful and efficient than monitoring wells in establishing the three-dimensional distribution of naphthalene in the Site 24 aquifer. Assuming the data to be accurate, since the MLS is able to take groundwater samples at smaller intervals, it is more effective than the monitoring well at determining the height of the plume. In one borehole the MLS is able to do the job of a well cluster; it can establish the degree and extent of contamination, as M-9 on transect C, M-11 on transect E, and M-1 on the centerline do for example, and establish the presence of contamination above and below clay lenses, as M-12 does on transect F and possibly M-2 on transect E.

Chapter 5

Conclusions and Recommendations

In general, the groundwater monitoring program incorporating both multi-level samplers and monitoring wells was successful. The multi-level samplers were effective at taking a number of groundwater samples in the vertical at one location, and our main goal of delineating the plume in three dimensions was met.

5.1 The Quality of the MLS Design Implemented at Site 24

The unique MLS design used at Site 24 that encorporated aluminum tubing and brass sampling ports suited its purpose well. The aluminum samplers performed better than the PVC samplers, which showed concentration levels consistently lower than a metal sampler only two feet away. The MLSs were convenient to construct, and hollow stem auger installation went fairly smoothly. The MLSs that collected the most useful data were built with 8 sampling ports and designed with an expectation of where the contamination was to be identified. It is possible that the glass wool was an unnecessary screening material. It could be responsible for some of the ports that did clog and the stainless steel screen might suffice in keeping soil out of the sample tubing.

This MLS design would not be appropriate for sites with low water tables. Peristaltic pumps will are not able to draw water from over 25 feet away, so a different design is required for such applications.

5.2 Reliability of Data Collection

The study done on duplicates of MLS samples supports the conclusion that groundwater samples from the MLS are reliable. Samples taken consecutively from the same MLS under the same conditions produced samples with concentrations within an average of 10% of each other. Concentrations from the same sampling port sometimes varied widely over time. The natural 10% variation and recharge events cannot explain concentration differences up to an order of magnitude. Errors must have been made during handling of the sample prior to analysis in order to produce such anomolous data points.

The literature supports the effectiveness of the sampling technique (peristaltic pump after sample vial) and MLS materials (chosen for their inertness towards organics) used. The tracer tests demonstrated that the MLSs were for the most part taking groundwater samples from the area local to each sampling port. More tracer tests should be carried out, however, to verify the absence of crosstalk between the many sampling ports in the site.

The MLS data, compared to adjacent monitoring well data, did not seem to consistently meet the higher values expected from more discrete samples. This may be due, however, to the differences in the plume over the distance between the samplers and wells. Special samplers could be designed that acted like both monitoring wells and MLSs to more closely compare groundwater samples from the two.

5.3 MLSs versus Monitoring Wells versus Soil Sampling

In general, the multi-level sampling technique was more effective than the monitoring well at determining the extent of the naphthalene plume at Site 24 in the vertical direction. Although MLSs require more samples, more information becomes available on the extent of the contamination. Because of the smaller amount of data analysis involved, monitoring wells were the most economical way to bound the plume, i.e. simply establish where the contamination is not in order to locate the plume edges. The combination of monitoring well clusters and MLSs, installed in several rounds, provided excellent data on the extent of the plume in the three dimensions. Data from MLSs alone would be weak in information in the transverse and longitudinal directions (assuming that the total number of samplers id limited). Data from monitoring wells alone would give little resolution of the plume in the vertical.

Soil data is very useful for detecting the presence of contamination and for siting the locations of well screens and MLS sampling ports. No relationship can be established however, that reliably correlates the values of naphthalene in the soil with naphthalene in the groundwater.

5.4 Where the MLS Technique is Best Applicable

The MLS technique is most effective when used at a sandy, relatively homogeneous site. The important criterion of proper caving of the aquifer material around the MLS occurs at such sites where consolidated soils are not encountered. It is possible to install MLSs through lenses of consolidated material with the use of bentonite grout to seal around the MLS at the level of the lens.

MLSs are a cost-effective way of collecting data to resolve the plume in the vertical dimension. Since MLSs are data-intensive and must be custom-made or specially ordered, they are primarily applicable to sites where three-dimensional data is important for tracking the plume or for modeling in three dimensions.

5.5 Areas of Future Study

Little is known in the area of multi-level sampling about the degree to which the disturbance of the zone around the MLS during installation affects the authenticity of the groundwater taken from each sampling port. Data profiles taken from the MLSs seem to indicate that samples are from different zones since they vary in a curve similar to what is expected. Tracer tests carried out at the Columbus site to look for signs of short-circuiting showed changes in the groundwater movement local to the MLS (TVA, 1988). It is possible that samples from MLSs represent water from different zones, but not necessarily the one zone at each level. Laboratory studies or computer models could be carried out to examine the flow patterns to a MLS during sampling under conditions of no disturbance and modeled disturbance (specific conductivity profiles). Special tracer tests could be designed to measure the hydraulic conductivity or porosity local to sampling ports in situ, and these values could be compared to values of undisturbed aquifer in that soil zone.

In most proposals to examine the reliability of the multi-level sampling technique we come up against a paradox. In order to take a groundwater sample representative of a particular location in space, the properties of the aquifer are unavoidably altered by the installation of the apparatus required to take the sample. The best an engineer can hope for is a minimization of the uncertainties involved in taking that measurement. Most of the controllable uncertainties encountered in multi-level sampling have been addressed in this thesis, such as materials selection, sampling protocol and installation. Uncontrollable uncertainties such as the natural heterogeneities present in aquifers make groundwater monitoring a challenge no matter what sampling well is used. At Site 24 uncertainties were minimized to such an extent that the data on the concentration of naphthalene in the groundwater was as authentic as data from any other groundwater monitoring technique.

Appendix A

Site 24 MLS Concentration Data

These data are the results of analyses performed by CAA of Boston, Mass. by the GC technique indicated (FID or PID). "ND" stands for non-detect, "lost" means the sample was taken but damaged before it could be analyzed, and "dry" means the port was clogged or not enough sample water was able to be withdrawn from the port. Some values are averages of duplicate samples, all of which are shown in Table 4-IV.

		Naphthalene concentrations (ppm)					
Well #	Label	6/6/89,FID	6/6/89,PID	9/5/89,FID	10/89,FID	4/26/90,FID	Elev. (ft)
P-1	11	dry		dry			287.21
	13	ND		dry			285.21
	15	ND		ND			283.21
	17	0.16		ND			281.21
	19	0.66	0.8	0.57			279.21
	21	0.895	1.07	1.21			277.21
	23	0.85		1.04			275.21
	25	dry		dry			273.21
M-1	11	dry		dry	dry		287.18
	13	lost		dry	dry		285.18
	15	0.08		0.023	ND		283.18
	17	lost		ND	0.06		281.18
	19	1.53	1.84	0.14	1.09		279.18
	21	1.59		1.44	1.58		277.18
	23	0.26		2.37	1.91		275.18
	25	lost		2.63	2.27		173.18
P-2	11	dry		dry			287.15
	13	dry		dry			285.15
	15	ND		dry			283.15
	17	lost		ND			281.15
	19	0.003		ND			279.15
	21	ND		ND			277.15
	23	ND		ND			275.15
	25	1.41	1.51	0.21			273.15
M-2	11	lost		ND	ND		278.93
	13	lost		ND	ND		276.93
	15	lost		ND	ND		274.93
	17	lost		0.011	ND		272.93
	19	lost		ND	0.04		270.93
	21	0.44		ND	0.06		268.93
	23	lost		ND	0.06		266.93
	25		0.91	ND	0.05		264.93
M-3	1				ND		282
	2				0.31		281
	3				2.43		280
	4				2.57		279
	5				1.98		278
	6				1.34		277
	7				2.01		276
	8				1.36		275
	9				1.67		274
	10				1.72		273

		Naphthalene concentrations (ppm)					
Well #	Label	6/6/89,FID	6/6/89,PID	9/5/89,FID	10/89,FID	4/26/90,FID	Elev. (ft)
M-4	1				1.45		280
	2				1.44		279
	3				1.93		278
	4				1.23		277
	5				1.32		276
	6				0.89		275
	7				0.69		274
	8				0.76		273
	9				0.45		272
	10				0.56		271
M-5	1				0.31		278.2
	2				0.20		277.2
	3				0.23		276.2
	4				0.29		275.2
	5				0.49		274.2
	6				0.50		273.2
	7				0.22		272.2
	8				0.14		271.2
	9				ND		270.2
	10				0.02		269.2
M-6	1				ND		276.2
	2				0.02		275.2
	3				0.07		274.2
	4				0.05		273.2
	5				0.07		272.2
	6				0.06		271.2
	7				0.14		270.2
	8				0.14		269.2
	9				0.13		268.2
	10				dry		267.2
M-7	1				ND		271.8
	2				dry		270.8
	3				ND		269.8
	4				ND		268.8
	5				ND		267.8
	6				ND		266.8
	7				ND		265.8
	8				ND		264.8
	9				ND		263.8
	10				ND		262.8

		Naphthalene concentrations (ppm)					
Well #	Label	6/6/89,FID	6/6/89,PID	9/5/89,FID	10/89,FID	4/26/90,FID	Elev. (ft)
M-8	1				ND		265.23
	2				dry		264.23
	3				0.11		263.23
	4				0.06		262.23
	5				0.45		261.23
	6				0.52		260.23
	7				0.52		259.23
	8				0.45		258.23
	9				0.39		257.23
	10				0.09		256.23
M-9	1					ND	281.8
	2					0.029	280.05
	3					0.248	278.3
	4					0.864	276.55
	5					0.763	274.8
	6					0.501	273.05
	7					0.682	271.3
	8					dry	269.55
M-10	1					dry	280.84
	2					dry	279.84
	3					0.252	278.84
	4					0.383	277.84
	5					0.459	276.84
	6					0.278	275.84
	7					0.189	274.84
	8					0.273	273.84
M-11	1					ND	279.07
	2					0.383	278.07
	3					0.975	277.07
	4					0.568	276.07
	5					0.769	275.07
	6					0.55	274.07
	7					0.361	273.07
	8					0.205	272.07
M-12	1					ND	270.5
	2					ND	269.5
	3					dry	265.92
	4					dry	265
	5					0.521	264.08
	6					0.336	263.17
	7					0.77	262.25
	8					1.277	261.34

Appendix B

Most Recent MLS Data Set

		Naphthalene concentrations (ppm)										
Well #	Label	ft below g	6/6/89,FID	6/6/89,PID	9/5/89, FID	10/89, FID	4/26/90, FI	6/25/90,FID	Elevation (ft	Gd Elev (ft	X-coord	Y-Coord
P-1	11	6	dry		dry				287.21	293.21	-10.14	18.06
	13	8	ND		dry				285.21			
	15	10	ND		ND				283.21			
	17	12	0.16		ND				281.21			
	19	14	0.66	0.8	0.57				279.21			
	21	16	0.895	1.07	1.21				277.21			
	23	18	0.85		1.04				275.21			
	25	20	dry		dry				273.21			
M-1	11	6	dry		dry	dry			287.18	293.18	-9.96	18
	13	8	lost		dry	dry		ND	285.18			
	15	10	0.06		0.023	ND		ND	283.18			
	17	12	lost		ND	0.06		ND	281.18			
	19	14	1.53	1.84	0.14	1.09		0.894	279.18			
	21	16	1.59		1.44	1.58		1.303	277.18			
	23	18	0.26		2.37	1.91		1.544	275.18			
	25	20	lost		2.63	2.27		1.998	173.18			
P-2	11	6	dry		dry				287.15	293.15	-24.72	27.24
	13	8	dry		dry				285.15			
	15	10	ND		dry				283.15			
	17	12	lost		ND				281.15			
	19	14	0.003		ND				279.15			
	21	16	ND		ND				277.15			
	23	18	ND		ND				275.15			
	25	20	1.41	1.51	0.21				273.15			
M-2	11	15	lost		ND	ND		ND	278.93	292.93	-39	51.66
	13	17	lost		ND	ND		ND	276.93			
	15	19	lost		ND	ND		ND	274.93			
	17	21	lost		0.011	ND		0.114	272.93			
	19	23	lost		ND	0.04		0.241	270.93			
	21	25	0.44		ND	0.06		0.351	268.93			
	23	27	lost		ND	0.06		0.333	266.93			
	25	29		0.91	ND	0.05		0.318	264.93			
M-3	1	12.5				ND		ND	283.03	295.53	11.15	-2.47
	2	13.5				0.31		ND	282.03			
	3	14.5				2.43		0.715	281.03			
	4	15.5				2.57		1.815	280.03			
	5	16.5				1.98		1.434	279.03			
	6	17.5				1.34		1.737	278.03			
	7	18.5				2.01		1.698	277.03			
	8	19.5				1.36		1.755	276.03			
	9	20.5				1.67		1.8	275.03			

Well #	Label	ft below g	6/6/89,FID	6/6/89,PID	9/5/89, FID	10/89, FID	4/26/90, FI	6/25/90,FID	Elevation (ft	Gd Elev (ft	X-coord	Y-Coord
	10	21.5				1.72		1.448	274.03			
M-4	1	14				1.45		1.693	280.49	294.49	6.02	6.18
	2	15				1.44		2.892	279.49			
	3	16				1.93		3.052	278.49			
	4	17				1.23		2.338	277.49			
	5	18				1.32		1.815	276.49			
	6	19				0.89		1.997	275.49			
	7	20				0.69		1.572	274.49			
	8	21				0.76		1.752	273.49			
	9	22				0.45		1.527	272.49			
	10	23				0.56		1.305	271.49			
M-5	1	15				0.31		0.013	278.16	293.16	-12.05	27.32
	2	16				0.20		0.04	277.16			
	3	17				0.23		0.296	276.16			
	4	18				0.29		0.5	275.16			
	5	19				0.49		0.436	274.16			
	6	20				0.50		0.487	273.16			
	7	21				0.22		0.665	272.16			
	8	22				0.14		0.279	271.16			
	9	23				ND		0.002	270.16			
	10	24				0.02		-	269.16			
M-6	1	16				ND		-	276.41	292.41	-25.28	42.75
	2	17				0.02		ND	275.41			
	3	18				0.07		ND	274.41			
	4	19				0.05		ND	273.41			
	5	20				0.07		ND	272.41			
	6	21				0.06		ND	271.41			
	7	22				0.14		ND	270.41			
	8	23				0.14		ND	269.41			
	9	24				0.13		ND	268.41			
	10	25				dry		-	267.41			
M-7	1	20.5				ND		ND	271.95	292.45	-36.4	54.26
	2	21.5				dry		ND	270.95			
	3	22.5				ND		ND	269.95			
	4	23.5				ND		ND	268.95			
	5	24.5				ND		ND	267.95			
	6	25.5				ND		ND	266.95			
	7	26.5				ND		ND	265.95			
	8	27.5				ND		ND	264.95			
	9	28.5				ND		0.028	263.95			
	10	29.5				ND		0.003	262.95			

MLS conc data

Well #	Label	ft below g	6/6/89,FID	6/6/89,PID	9/5/89, FID	10/89, FID	4/26/90, FI	6/25/90,FID	Elevation (ft	Gd Elev (ft	X-coord	Y-Coord
M-8	1	28				ND		ND	265.17	293.17	-47.5	74.1
	2	29				dry		0.17	264.17			
	3	30				0.11		0.159	263.17			
	4	31				0.06		0.13	262.17			
	5	32				0.45		0.27	261.17			
	6	33				0.52		0.453	260.17			
	7	34				0.52		0.484	259.17			
	8	35				0.45		0.574	258.17			
	9	36				0.39		0.577	257.17			
	10	37				0.09		0.558	256.17			
M-9	1	11					ND	ND	281.8	292.8		
	2	12.75					0.029	0.07	280.05			
	3	14.5					0.248	0.527	278.3			
	4	16.25					0.864	0.95	276.55			
	5	18					0.763	1.26	274.8			
	6	19.75					0.501	0.865	273.05			
	7	21.5					0.682	1.252	271.3			
	8	23.25						0.999	269.55			
M-10	1	11.66						0.012	280.84	292.5		
	2	12.66						ND	279.84			
	3	13.66					0.252	0.006	278.84			
	4	14.66					0.383	-	277.84			
	5	15.66					0.459	0.023	276.84			
	6	16.66					0.278	0.058	275.84			
	7	17.66					0.189	0.161	274.84			
	8	18.66					0.273	0.176	273.84			
M-11	1	15.33					ND	0.65	279.07	294.4		
	2	16.33					0.383	0.869	278.07			
	3	17.33					0.975	1.104	277.07			
	4	18.33					0.568	1.279	276.07			
	5	19.33					0.769	1.147	275.07			
	6	20.33					0.55	1.213	274.07			
	7	21.33					0.361	1.319	273.07			
	8	22.33					0.205	0.834	272.07			
M-12	1	23.5					ND	ND	270.5	294		
	2	24.5					ND	ND	269.5			
	3	28.08						ND	265.92			
	4	29						0.582	265			
	5	29.92					0.521	0.458	264.08			
	6	30.83					0.336	0.831	263.17			
	7	31.75					0.77	0.895	262.25			
	8	32.66					1.277	1.154	261.34			

Well #	Label	ft below g	6/6/89,FID	6/6/89,PID	9/5/89, FID	10/89, FID	4/26/90, FI	6/25/90,FID	Elevation (ft	Gd Elev (ft	X-coord	Y-Coord
M-13	1	13.33						0.018				
	2	15.83										
	3	18.33										
	4	20.83						0.123				
	5	23.33						0.387				
	6	27.33						0.023				
	7	28.33						ND				
	8	29.33						-				
M-14	1	9.58						ND				
	2	10.83						ND				
	3	12.08						ND				
	4	13.33						0.009				
	5	14.58						0.003				
	6	15.83						0.113				
	7	17.08						-				
	8	18.33						0.536				
M-15	1	21.5						0.338				
	2	22.5						0.223				
	3	23.5						-				
	4	24.5						0.207				
	5	25.5						0.19				
	6	26.5						0.207				
	7	30.67						0.013				
	8	31.67						0.012				
M-16	1	25.25						-				
	2	26.33						ND				
	3	27.42						0.095				
	4	28.5						0.336				
	5	29.58						0.91				
	6	30.67						0.879				
	7	31.75						1.083				
	8	32.83						0.413				

Appendix C
Well Locations and Head Data

								GW Elev (ft	GW Elev	GW Elev	GW Elev	GW Elev	GW Elev	GW Elev	GW Elev
Well ID	X-coord	Y-coord	Gnd-Elev		TOScreen	BOscreen		6/17/88	7/22/88	10/31/88	3/30/89	4/11/89	5/12/89	5/9/90	Jul-90
MW1	25.74	-4.32	296.99		290.99	285.99			288.52				288.92	290.36	289.7
MW2	25.62	-4.08	297.03		287.03	282.03			287.19				288.79	290.39	289.66
MW3	25.68	-3.66	296.83		282.83	277.83			287.14	286.27			288.78	290.31	289.55
MW4	25.44	-31.98	296.86						290.46	290.03			292.18		292.58
MW5	19.86	6.66	295.66		283.16	278.16			286.79	285.91			288.34	289.73	289.26
MW6	16.98	4.2	296.6		284.1	279.1			286.79	285.84			288.36	289.65	289.23
MW7	16.5	4.5	296.61		289.61	284.61			286.89	285.75			288.41	289.68	289.11
MW8	-0.9	11.1	293.5		277.5	272.5			284.49	284.03	284.88		286.69	287.65	286.9
MW9	-0.54	11.76	293.44		282.44	277.44			284.83	284.35	285.20		286.95	288.12	287.57
MW10	29.64	22.74	294.88		288.88	283.88			286.80	285.90			288.31		289.55
MW11	-2.88	4.32	294.27		268.27	263.27			282.24	281.88	282.52		283.77		283.65
MW12	-32.1	34.08	292.12		276.12	271.12			280.78	280.67	281.76		283.35	283.86	283.58
MW13	12	-1.26	294.33		279.33	277.33			286.46	285.71			288.30	289.43	289.33
MW14	34.2	-12.6	297.99		289.99	287.99				288.02			289.41	291.34	290.71
MW15	12.3	-1.26	294.54		288.04	285.04			287.10	286.09			- -	289.21	289.21
MW16	-32.34	34.8	292.01		278.01	276.01			281.67	281.13	281.87		283.45	283.91	283.61
MW17	-31.8	34.68	292.11		283.11	280.11			281.67	281.11	281.99		283.62	283.96	283.63
MW19	34.26	-13.02	297.83		280.83	275.83			287.72	286.86			289.56		291.05
MW20	2.58	18.54	293.32		276.32	271.32			285.06	284.37	285.10		286.76		287.37
MW21	-2.76	4.74	294.16		284.16	281.16			285.20	284.40	285.10		286.92		287.44
MW22	-1.8	4.2	294.45		273.45	271.45			282.85	282.56	282.01		284.32		284.4
MW23	-59.82	49.86	293.39		281.39	276.39			278.38	277.70	278.55		280.00		279.75
MW24	-0.06	12	293.31		287.31	282.31			285.30	284.41	285.21		286.98	287.92	287.49
MW25	10.98	-1.26	294.26		281.76	279.26			286.78	285.84			- -	289.28	288.75
MW26	-59.34	51.96	293.23		266.23	261.23			- -	- -			275.54		274.94
MW27	-46.5	78.06	293.18		260.68	258.18			- -	- -			271.75	271.28	271.88
MW28	-46.38	77.76	293.21		265.71	263.21			- -	- -			271.96	271.83	271.51
MW29	-37.92	93.42	293.64		264.64	259.64			- -	- -			269.95		268.82
MW30	-48	40.26	294.35		271.35	266.35			- -	- -			267.08		297.39
MW31	-38.58	52.62	293.06		269.56	267.06			- -	- -			279.60	279.64	279.86
MW32	-38.64	52.2	292.99		272.49	269.99			- -	- -			279.69	279.97	279.59
MW33	-28.8	63.12	291.28		266.28	261.28			- -	- -			277.77		277.47
MW34	-16.32	52.8	292.36		277.36	272.36			- -	- -			283.00		283.82
MW35	-30	13.8	292.87		274.87	269.87			- -	- -			284.73		285.23
MW36	-13.8	26.04	293.25		272.75	270.25			- -	- -			285.54	286.16	285.94
MW37	-13.44	26.22	293.23		275.73	273.23			- -	- -			285.66	286.30	286.16
MW38	-6.3	30.36	292.76		274.76	269.76			- -	- -			283.47		284.74
MW39	25.8	14.4	295.45		286.45	281.45			- -	- -			288.34		289.22
MW40	7.92	-8.76	295.18		282.18	277.18			- -	- -			288.16		288.94
MW41	-10.44	-2.7	294.18		278.68	273.68			- -	- -			287.15		287.73
MW42	-40.62	25.08	292.11		277.11	272.11			- -	- -			283.33		283.2
MW43	-37.14	28.44	291.68		276.68	271.68			- -	- -			283.54		283.83
MW44	23.82	-2.46	296.89		284.5	282								289.93	289.96
MW45	24.6	-2.65	296.73		282.5	280								290.53	290.58
MW46	-12.95	26.15	293.15		280.41	278.01								286.01	285.68
GW100	37.32	-10.2	297.4					285.89	- -	285.73			289.43		297.61

Well_ID	X-coord	Y-coord	Gnd-Elev		TOScreen	BOscreen		6/17/88	7/22/88	10/31/88	3/30/89	4/11/89	5/12/89	5/9/90	Jul-90
HW0	32.46	-1.14	295.41					287.57	- -	- -		287.56	288.99		290.37
HW50	32.34	-5.52	296.37					285.30	- -	- -		287.50	289.11	290.53	290.17
IE0	27.48	4.02	296.09					287.37	287.11	- -		287.35	288.83	290.18	289.71
IE50	27.54	8.46	295.66						287.00	- -		287.13	288.53	289.98	289.48
IE100	27.6	13.14	295.89					285.39	287.73	285.32		287.08	288.44		289.33
IE200	26.1	25.8	294.28						286.25	- -		286.55	287.78		288.53
IW0	27.6	-1.2	294.99					287.27	287.14	286.18		287.31	- -	290.32	289.79
IW50	27.6	-4.86	296.15					285.76	287.38	286.69		287.58	288.99	290.55	290.1
IW100	27.6	-10.14	296.46					286.95	287.43	286.50		287.65	288.99		291.69
IW150	27.66	-15.18	298.18					287.85	287.46	- -		287.79	289.28		
JE0	22.56	3.6	295.83					286.99	286.59	- -		287.03	288.22		289.38
JE100	21.54	14.94	295.45					283.74	286.78	- -		286.94	288.24		289.09
JE150	21.54	19.5	294.91						286.61	- -		286.85	288.15		288.93
JE200	21.36	22.14	294.9						286.27	285.40		286.59	287.83		288.55
JW0	22.44	-1.38	295.33					287.26	287.07	286.18			288.89		295.86
JW50	22.68	-5.34	295.88					285.14	286.75	285.02			288.78		289.59
JW100	22.8	-10.2	296.35					285.69	287.29	286.95			288.97		
JW250	23.64	-25.2	297.29					286.89	287.54	- -		288.02	289.58		290.81
KE0	17.4	3.54	295.76					287.23	287.04	- -			- -		289.16
KE50	17.58	8.28	295.9					285.73	- -	285.83		286.97	- -		289.06
KE100	15.84	15.06	295.64					285.04	- -	- -			287.97		288.67
KE300	13.08	34.8	293.96						285.52	284.78		285.96	287.14		287.39
KW0	17.46	-1.2	295.01					287.27	287.06	285.96			- -	289.84	289.3
KW50	18.12	-5.04	296.57					286.20	286.87	- -			288.47	289.96	289.42
LE0	12.6	3.6	295.06					286.65	286.27	- -		286.46	288.19		288.96
LE25	12.42	5.7	295.52					283.71	- -	285.38		286.66	287.98	289.38	288.77
LE50	12.48	7.98	295.45					283.97	286.41	285.36		286.69	287.93	289.09	288.63
LE192	12.84	22.2	294.04					282.98	285.80	- -		286.26	287.41		287.98
LW100	13.32	-10.8	296.42					286.25	286.62	285.80		287.12	288.52	289.91	289.47
ME0	8.1	3.6	294.39					285.60	285.72	- -		285.94	287.65	288.35	287.92
ME25	8.1	5.58	294.41					284.54	286.10	285.02		286.43	287.65	288.78	288.3
ME50	7.62	8.04	293.94					284.10	285.97	285.01		286.38	287.60	288.66	288.24
ME100	7.68	13.68	293.67					283.53	285.71	284.80		286.16	287.32		287.92
MW0	7.44	-1.26	294.34					283.90	286.83	- -		286.91	287.59	289.41	288.93
MW100	8.3	-9.9	295.98					285.32	286.36	285.46		286.90	288.15	289.49	288.98
NW0	2.4	-1.2	294.08						283.09	- -		286.65	287.53	289.05	288.54
NW50	3	-6.42	297.11						288.08	287.38		288.56	287.96		290.61
OE100	-2.82	13.32	293.67					282.19	- -	- -		285.66	286.79		287.31
OE200	-3.24	22.98	293.37					282.28	284.48	- -	284.49	285.11	286.31		286.74
OW0	-2.4	-1.38	293.58					282.20	285.88	- -		286.45	287.40		288.26
OW100	-3	-11.1	293.2						286.10	- -			288.00		288.64
PS10	-40.8	120.9	277.22						- -	- -	273.15	273.69	274.34		273.13
PS11	-43.08	115.08	277.6						- -	- -	274.59	274.09	274.70		273.63
PS12	-45.6	109.26	277.7						- -	- -	274.01	274.23	274.86		274.04
PS13	-47.7	103.98	277.62						- -	- -	274.10	274.25	275.00		274.29
PS14	-49.86	98.46	277.45						- -	- -	274.09	274.23	275.03		274.39
PS15	-52.14	94.62	277.28						- -	- -	274.30	274.39	275.09		274.58

Well_ID	X-coord	Y-coord	Gnd-Elev		TOScreen	BOscreen		6/17/88	7/22/88	10/31/88	3/30/89	4/11/89	5/12/89	5/9/90	Jul-90
PS16	-54	88.2	277.56						- -	- -	275.12	275.02	275.82		275.34
PS17	-56.6	82.5	277.63						- -	- -	276.02	275.72	276.52		276.1
PS18	-67.06	76.8	277.81						- -	- -	276.27	275.95	276.43		277.32
PS19	-60.6	71.28	277.28						- -	- -	276.60	276.30	276.58	276.63	276.63
PS20	-63.6	66	277.58						- -	- -	277.13	276.85	277.20		277.15
PS21	-66.9	60.72	277.64						- -	- -	277.25	276.85	277.20		277.01
PS22	-70.66	56.1	277.83						- -	- -	277.45	277.00	277.23		277.05
PS23	-75.72	52.26	277.43						- -	- -	276.81	276.53	276.75		276.58
PS24	-73.56	87.72	268.89						- -	- -			267.07		271.44
PS25	-71.04	93.78	264.04						- -	- -			262.46		262.07
PS26	-70.98	95.04	258.74						- -	- -			258.74		258.74
PS27	-70.08	93.3	263.15						- -	- -			262.68		262.41
PS28	-65.94	106.8	267.96						- -	- -			266.56		265.51
PZ101	-32.76	118.08	279.94						- -	- -			274.29		273.26
PZ102	-80.04	83.46	268.05						- -	- -			265.50		263.51
PZ103	-85.2	91.2	266.95						- -	- -			259.70		259.12
PZ104	-77.7	106.8	266.04						- -	- -			260.24		259.44
PZ105	-5.34	93.6	274.07						- -	- -			268.02		
PZMW10	29.46	23.22	294.38						287.09	285.86		287.07	- -		289.25
QE200	-12.6	25.98	293.19						- -	- -	283.17	283.77	- -		286.24
QW0	-12.24	-1.38	293.4						- -	- -		286.03	- -		287.69
SP	-60.48	20.7	282.99										281.96		281.53
SW100	-32.34	-9.42	293.6						- -	- -			- -		286.8
XC6	-35.64	9.6	292.93						- -	- -		283.63	284.60		284.57
XC7	-26.46	16.2	292.67						- -	- -		283.78	284.98		284.96
XC8	-18.54	22.62	293.25						- -	- -		284.24	285.52		
XC9	-10.5	27.12	292.44						- -	- -	283.11	283.64	285.74		
XC10	-7.2	31.2	292.76						- -	- -	283.86	284.44	285.79	286.29	284.89
XD27	-16.06	54.18	292.02						- -	- -		281.45	282.98		282.93
XD28	-21.36	47.64	292.3						- -	- -		279.00	283.26		283.13
XD33	-42.18	23.1	292.26												283.54
XD34	-37.14	28.8	291.62						- -	- -			283.61	283.91	283.43
XD35	-41.4	23.4	292.2						- -	- -			283.25		283.51
XE1	-26.1	67.14	291.54						- -	- -		277.75	279.22		278.71
XE2	-31.5	60	291.78						- -	- -	279.67	280.03	281.27	281.25	280.9
XE3	-38.1	52.2	292.5						- -	- -	277.27		- -		279.25
XE4	-55.14	31.2	295.58						- -	- -		280.71	281.67		281.38
XE5	-47.52	39.7	293.76						- -	- -	278.55	278.81	281.65		281.05
XE29	-43.08	44.94	294.14						- -	- -			280.96	281.25	280.77
XF16	-53.04	63.48	293.67						- -	- -	277.59	277.83	279.17	279.23	277.66
XF17	-48.9	72.48	294.07						- -	- -	276.93	277.20	278.61		278.36
XF19	-42.18	85.14	289.77						- -	- -		275.82	277.12		
TAB1	-8.82	-11.4	294.73												288.23
TBC1	-25.2	3.6	293.78												286.62
TBC2	-12	6.42	293.42												287.32
TBC3	-13.92	16.08	293.54												285.65
TCD2	-28.14	25.08	292.79												284.47

Well ID	X-coord	Y-coord	Gnd-Elev		TOScreen	BOscreen		6/17/88	7/22/88	10/31/88	3/30/89	4/11/89	5/12/89	5/9/90	Jul-90
TCD3	-21.24	27.9	292.8												285.1
TCD4	-16.98	35.34	292.92												285.14
TDE1	-42.84	34.66	292.66												294.4
TDE2	-37.18	39.3	292.66												282.3
TDE3	-33.18	47.64	292.28												261.77
TDE4	-25.8	56.52	292												281.77
CLS1	-0.3	88.8	265.42										265.42		
CLS2	-1.2	92.1	262.8										262.60		
CLS3	-1.08	96	255.39										255.39		
CLS4	-3	101.1	253.91										253.91		
CLINE	-82.44	75.42	252.77										252.77		

Appendix D

Monitoring Well Concentration Data

Concs.- Date order

Sta.	Well#	7/1/88 FID	7/20/89GC	11/88 GC	11/88 FID	6/89 FID	8/89 FID	10/89 FID	Technique	Company	Gd. Elev (ft)	Scrn Elev-T	Scrn Elev-B	Screen Bo	Screen Top
1	1		0.52						SS bail	CAA	296.99	290.99	285.99	11	6
1	1		0.35						PVC bail	CAA				11	6
1	1					0.002			Pump	CAA				11	6
1	2	3.65							Pump (20)	Dan	297.03	287.03	282.03	15	10
1	2		5.1	3.30	4.307		3.182		SS bail	CAA				15	10
1	2			1.4	1.674	1.35	1.997		Pump	CAA				15	10
1	2			0.82	0.854				Pump+filt	CAA				15	10
1	3		3.3		2.099				SS bail	CAA	296.83	282.83	277.83	19	14
1	3					1.8			Pump	CAA				19	14
2	10		0		0				SS bail	CAA	294.88	288.88	283.88	11	6
2	10					0			Pump	CAA				11	6
3	6	2.53							bail	Dan	296.6	284.1	279.1	17.5	12.5
3	6	2.145							Pump (120)	Dan				17.5	12.5
3	6	2.78							Pump (950)	Dan				17.5	12.5
3	6		1.2	0	0.003				Pump+Filt	CAA				17.5	12.5
3	6		0.88			0.77			Pump	CAA				17.5	12.5
3	6			0.061	0.061				SS bail	CAA				17.5	12.5
3	7		0.09		0.001				SS bail	CAA	296.61	289.61	284.61	12	7
3	7		0.13						PVC bail	CAA				12	7
3	7					0.03			Pump	CAA				12	7
4	5		0.00		0.189				SS bail	CAA	295.66	283.16	278.16	17.5	12.5
4	5				0.098				Pump+filt	CAA				17.5	12.5
4	5					0.02			Pump	CAA				17.5	12.5
5	13		2.4	3.1	4.251		2.15		SS bail	CAA	294.33	279.33	277.33	17	15
5	13		1.9						PVC bail	CAA				17	15
5	13			2.8		0.86	4.00		Pump	CAA				17	15
5	13			2.6	3.337				Pump+filt	CAA				17	15
5	15		0	0					SS bail	CAA	294.54	288.04	285.04	9.5	6.5
5	15					0			Pump	CAA				9.5	6.5
5	25		2.1						Pump+Filt	CAA	294.26	281.76	279.26	15	12.5
5	25		1.9			0.34			Pump	CAA				15	12.5
5	25				3.104				SS bail	CAA				15	12.5
6	20		0	0	0				SS bail	CAA	293.32	276.32	271.32	22	17
6	20			0					Pump+filt	CAA				22	17
6	20					0.003			Pump	CAA				22	17
7	8		0.022		0.84				SS bail	CAA	293.5	277.5	272.5	21	16
7	8					0.27			Pump	CAA				21	16
7	9	2.8							bail	Dan				16	11
7	9	2.747							Pump (130)	Dan	293.44	282.44	277.44	16	11
7	9		1.2	0.925	1.629		1.35		SS bail	CAA				16	11
7	9			0.67	0.717	3.4	1.80		Pump	CAA				16	11
7	9			1.01	1.177				Pump+filt	CAA				16	11
7	24		0.225		1.046				SS bail	CAA	293.31	287.31	282.31	11	6
7	24					0.2			Pump	CAA				11	6

Sta.	Well	7/1/88 FID	7/20/88GC	11/88 GC	11/88 FID	6/89 FID	8/89 FID	10/89 FID	Technique	Company	Gd. Elev (ft)	Scrn Elev-T	Scrn Elev-B	Screen Bo	Screen Top
8	11		0	0					SS bail	CAA	294.27	268.27	263.27	31	26
8	11					0			Pump	CAA				31	26
8	21		0.083	0.186					SS bail	CAA	294.16	284.16	281.16	13	10
8	21					0.01			Pump	CAA				13	10
8	22		0.006	0.001					SS bail	CAA				23	21
8	22		0.009						PVC bail	CAA	294.45	273.45	271.45	23	21
8	22					0.01			Pump	CAA				23	21
9	12		0.011	0.74	0.925		0.92		SS bail	CAA	292.12	276.12	271.12	21	16
9	12			1.06	1.174				Pump+filt	CAA				21	16
9	12			1.1	1.244	0.39	1.01		Pump	CAA				21	16
9	16		0.24						Pump+Filt	CAA	292.01	278.01	276.01	16	14
9	16		0.13			0.41	1.10		Pump	CAA				16	14
9	16				0.052		0.80		SS bail	CAA				16	14
9	17		0	0					SS bail	CAA	292.11	283.11	280.11	12	9
9	17					0			Pump	CAA				12	9
10	14					0			Pump	CAA	297.99	289.99	287.99	10	3
10	19		0	0					SS bail	CAA	297.83	280.83	275.83	22	17
10	19		0						PVC bail	CAA				22	17
10	19					0			Pump	CAA				22	17
11	23		0		0				SS bail	CAA	293.39	281.39	276.39	17	12
11	23					0			Pump	CAA				17	12
11	26					0			Pump	CAA	293.23	266.23	261.23	32	27
12	29					0			Pump	CAA	293.64	264.64	259.64	34	29
13	27					0.45	0.113		bail	CAA	293.18	260.68	258.18	35	32.5
13	27						0.407		Pump	CAA					
13	28					0.065	0.064		Pump	CAA	293.21	265.71	263.21	30	27.5
13	28					0.028	0.046		bail	CAA				30	27.5
14	38					0			Pump	CAA	292.76	274.76	269.76	23	18
15	31					0.28	0.285		Pump	CAA	293.06	269.56	267.06	26	23.5
15	31					0.21	0.18		bail	CAA				26	23.5
15	32					0	0.005		Pump	CAA	292.99	272.49	269.99	23	20.5
	32						0.016		Bail	CAA					
17	34					0			Pump	CAA	292.36	277.36	272.36	20	15
18	35					0			Pump	CAA	292.87	274.87	269.87	23	18
19	36					1.1	1.3		Pump	CAA	293.25	272.75	270.25	23	20.5
19	36					1.1	1.125		bail	CAA				23	20.5
19	37					1.2	1.635		Pump	CAA	293.23	275.73	273.23	20	17.5
19	37					1.4	1.305		bail	CAA				20	17.5
19	46							0.342	Pump	CAA	293.21	280.41	278.01	15.2	12.8
19	46							0.263	bail	CAA				15.2	12.8
20	33					0			Pump	CAA	291.28	266.28	261.28	30	25
21	39					0			Pump	CAA	295.45	[illegible]6.45	281.45	14	9
22	40					0			Pump	CAA	295.18	[illegible]2.18	277.18	18	13
23	41					0			Pump	CAA	294.18	278.68	273.68	20.5	15.5
24	42					0			Pump	CAA	292.11	277.11	272.11	20	15
25	43					0.31			Pump	CAA	291.68	276.68	271.68	20	15

Sta.	Well	7/1/88 FID	7/19/88MS	11/88MS	11/88 FID	6/89 FID	8/89 FID	10/89 FID	Technique	Company	Gd. Elev (ft)	Scrn Elev-T	Scrn Elev-B	Screen Bo	Screen Top
	44							4.025	Pump	CAA	296	284.5	282	14	11.5
	44							3.265	bail	CAA				14	11.5
	45							1.825	Pump	CAA	296	282.5	280	16	13.5
	45							1.845	bail	CAA				16	13.5

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