

DRAFT REPORT

Evaluation of Available Remedial Technologies and Conceptual Design of Recommended Remedial Approach for the Person Generating Station, Public Service Company of New Mexico

Prepared For

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ES
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EXECUTIVE SUMMARY

The objective of this report is two-fold. First, this report documents the screening of technologies and development of a remedial approach appropriate for use at the Person Generating Station site to remove volatile organic compound (VOC) contamination in both soil and shallow groundwater. Second, the report also presents results from predictions of the shallow groundwater VOC plume migration and associated risks at the site using several different response action scenarios and a fate and transport model based on available site data.

Earlier investigations indicate little potential for dense nonaqueous phase liquid (DNAPL) contamination at the site. Site monitoring activities have revealed concentrations of several volatile organic contaminants, including 1,1-dichloroethene (1,1-DCE), 1,1,1-trichloroethane (1,1,1-TCA), and tetrachloroethene (PCE), in both soil and shallow groundwater. The source of this contamination was a below-grade waste oil tank, which was removed from service in October 1983. The tank was used to store a variety of liquid waste streams, including steam cleaning residues containing chlorinated solvents.

A range of remedial technologies appropriate for use at sites with soil and groundwater VOC contamination are briefly described and then evaluated against OSWER Directive 9902.3 criteria. Selected technologies were then combined into a two-phased remedial approach recommended for the Person Generating Station site. The recommended remedial approach makes use of several technologies considered appropriate for the site, namely, groundwater pumping, above-ground treatment of groundwater using air stripping technology, and soil vapor extraction for the removal of VOCs from source area soils. A two-phased approach is recommended which includes an initial phase of pilot testing followed by a final design and full-scale remediation phase.

The recommended remedial action should also account for natural fate and transport processes occurring at the site. A three-dimensional fate and transport model, MODFLOW/MT3D, was used to predict possible shallow groundwater VOC plume behavior in the event that no action was taken at the site. The model was then used to simulate the impact of both nine years of pumping and removal using five extraction wells and 11 years of natural attenuation and six years of pumping and removal using four extraction wells and 14 years of natural attenuation. All simulated data is presented at time equal to 20 years from initiation of treatment. Fate and transport data derived from the no action model run indicate that the areal extent of the contaminant plume should change very little over the next twenty years. Additionally, it appears that minimal downgradient or vertical migration is to be expected under pumping or

non-pumping alternatives. A significant decrease in the concentration of VOCs in the shallow groundwater is predicted following implementation of pump-and-treat technologies. The model demonstrates that VOC concentrations could be reduced below 5 ppb after only six years of pumping using four extraction wells and 14 years of natural attenuation.

Data from these modeling efforts were then coupled with site characteristics to estimate the potential risk of human exposure to contaminants from the Person Generating Station site. Under all pump-and-treat options considered at this site, there are no completed pathways of exposure from any of the contaminated media present at the site. The lack of significant downward or vertical migration of any remaining VOCs in the shallow groundwater has effectively isolated the contamination from potential human and ecological receptors. Therefore, no current risks to human health or the environment are anticipated to exist from the shallow groundwater contamination. Implementation of soil gas vapor extraction techniques at the site during remediation activities may require offgas treatment to minimize potential exposure of site workers to contamination. Future risks posed by remaining contaminants should be assessed after evaluating the results of soil vapor extraction and groundwater pump-and-treat activities.

SECTION 1

INTRODUCTION

This report provides supporting data necessary to prepare a remedial action plan for the Person Generating Station site shallow groundwater remediation project as directed in Phase II, Item 1.B, of the Corrective Action Directive (CAD).

1.1 PURPOSE AND ORGANIZATION OF REPORT

Engineering-Science, Inc. (ES) was contracted by the Public Service Company of New Mexico (PNM) to perform several critical tasks involved in selecting, designing, and implementing an appropriate response action at the Person Generating Station site which is located near Albuquerque, New Mexico. This report documents the completion of two of these assigned tasks: (1) identification, evaluation, and recommendation of appropriate remedial technologies, which includes a preliminary investigation of the effect of natural fate and transport processes on shallow groundwater volatile organic compound (VOC) plume behavior and possible exposure potential at the site, and (2) a conceptual design of the recommended remedial approach.

This report is organized to clearly document the development of the recommended remedial alternative. Background information, including historical information and a summary of the nature and extent of contamination at the site, provides a basis for identifying the range of alternatives to consider for implementation. Several remedial technologies, which are effective in removing VOC contamination, are described briefly. A series of remedial technology evaluation criteria are used to identify promising technologies for the Person Generating Station site. Technologies surviving this screening process are incorporated into a final recommended remedial alternative for the site, which is to be implemented in two distinct phases.

The recommended groundwater remedial alternative--a combined source removal and pump-and-treat approach--is then investigated more fully with respect to long-term effectiveness and required duration to achieve desired cleanup levels. A three-dimensional fate and transport model based on available site data is used to investigate the possible behavior of the shallow groundwater VOC plume under three different groundwater remediation scenarios. Because of physical and chemical limitations, pump-and-treat technologies will not remove all of the VOC contamination in the shallow groundwater. Some level of groundwater contamination, which depends on the nature and duration of pump-and-treat activities implemented at the site, will remain in the aquifer to naturally attenuate through dispersion, hydrolysis, biodegradation,

sorption, and volatilization into the vadose zone. The groundwater flow model MODFLOW was coupled to the contaminant transport model MT3D to predict the position and concentrations in the plume following 9 years of pump-and-treat activities using 5 wells, and 6 years of pump-and-treat activities using 4 wells. All model data are calibrated using recent groundwater data. The results from these modeling runs are used to identify any potential migration to receptors, and to estimate the potential risk to human health and the environment from implementing any of the remedial alternatives under consideration.

1.2 BACKGROUND INFORMATION

The following sections describe the history of the site leading up to and following the discovery of the VOC contamination and relevant site characteristics, including the nature and extent of VOC contamination in both soil and shallow groundwater media.

1.2.1 Site History and Description

The specific characteristics of the Person Generating Station site, including source of contamination and other relevant physical aspects of contaminated media, will drive the identification and ultimate selection of appropriate remedial technologies and supporting fate and transport modeling.

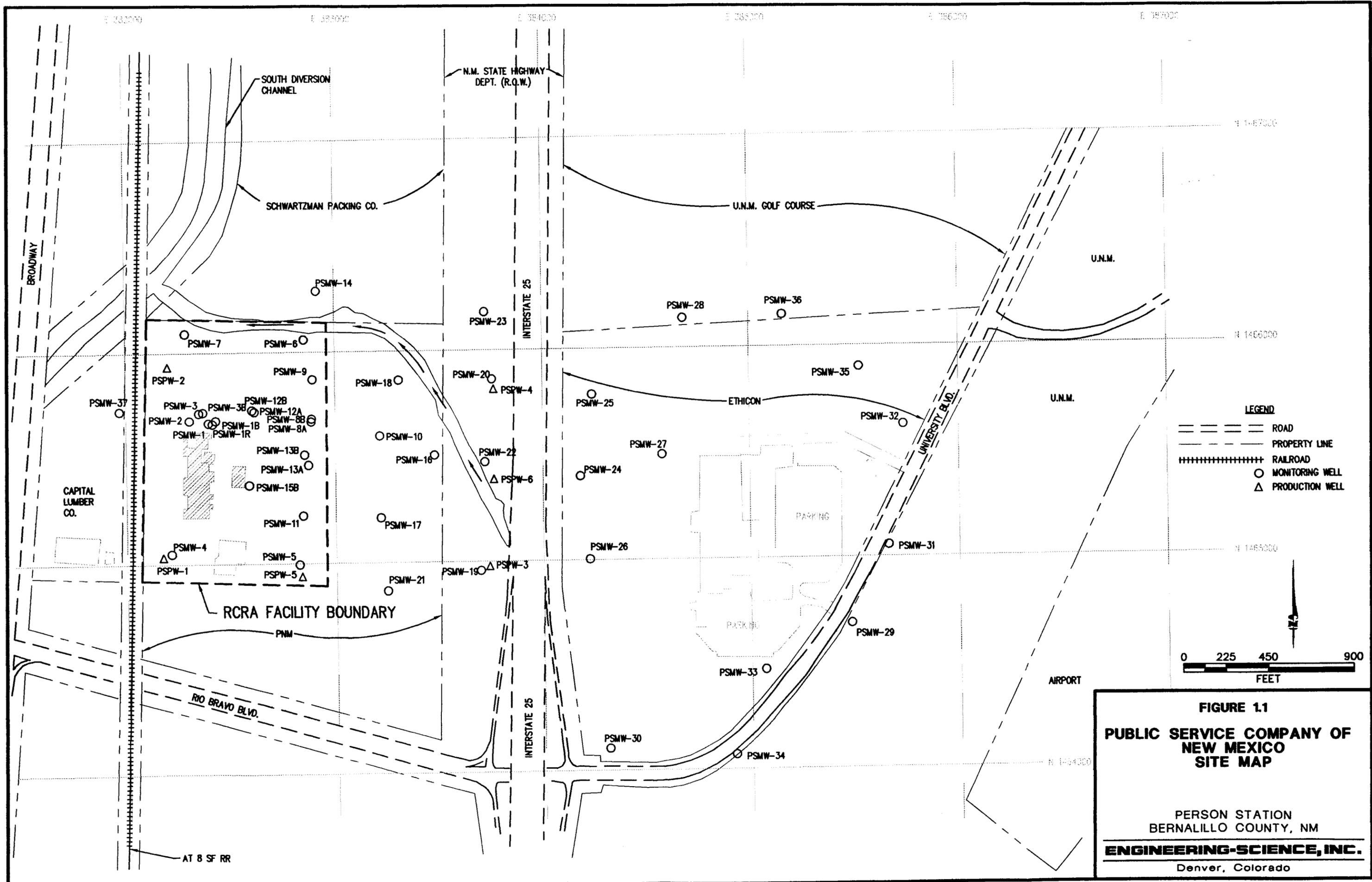
1.2.1.1 Operational History

The Person Generating Station site, which was operated and maintained by the Public Service Company of New Mexico (PNM), is located in the Albuquerque Basin, a physiographic drainage basin in the middle part of the long Rio Grande Valley which extends northward through the length of New Mexico (Kelley, 1977). Interstate 25 is located approximately 1000 feet to the east (see Figure 1.1).

The Person Generating Station site included a maintenance area to support, among other activities, equipment cleaning efforts. The parts wash area included a sump and a below-grade, vertically-placed 3.5' x 10' cylindrical waste oil storage tank located on the north side of the site to collect wastes generated during equipment cleaning.

Liquid wastes collected in the sump were piped approximately nine feet to the below-grade waste oil tank. Historical records and interviews with retired personnel indicate that waste oils and greases, kerosene, a water-trisodium phosphate mixture used in steam cleaning, Stoddard Solvent, Dowclene EC, and other solvent mixtures generated during maintenance activities were piped into the tank for storage (METRIC, 1993). Dowclene EC is a generic solvent with two primary active ingredients: 1,1,1-trichloroethane (1,1,1-TCA) and tetrachloroethene (PCE). Records suggest that major use of the Dowclene EC product began in 1979. Equipment repainting activities conducted in 1980 generated a new type of liquid effluent, including waste paint, paint thinners, and turpentine, that also was collected in the waste oil tank. Maintenance personnel noted when the tank appeared to be full and arranged for various waste oil reclaimers to remove the contents and recycle the material at other locations.

The tank was apparently in use from about July 1976 until October 13, 1983, when it was discovered that the tank lacked an impermeable bottom (i.e., the tank bottom



was constructed of soil). Upon discovery of this information on October 13, 1983, PNM immediately emptied the tank and removed it from service. PNM notified the U.S. Environmental Protection Agency (USEPA), the New Mexico Environmental Improvement Division (the predecessor agency to the New Mexico Environment Department, NMED), and the National Response Center of the discovery. PNM arranged for the most highly contaminated source material to be removed from the bottom of the tank and placed in 55 gallon steel drums in 1983; this drummed material was ultimately transported offsite for disposal as hazardous waste in 1987.

Following removal of the tank from service, PNM installed a closure cap on the 25' x 35' source area to minimize infiltration. The cap was comprised of a minimum 6 inch thick concrete cap over a minimum 6 inch thick layer of compacted soil over two layers of 80 mill High Density Polyethylene (HDPE) plastic sheeting. The excavated material from the tank area was replaced with gravel overlain by compacted soil.

In an effort to assess the potential environmental contamination stemming from the use of this waste oil tank, PNM has conducted two assessment projects. The first assessment, initiated immediately after discovery, was completed in late 1985. Data from this assessment were used to develop the Closure Plan and the existing Post-Closure Care Resource Conservation and Recovery Act (RCRA) permit. The second assessment was conducted pursuant to the CAD, which was issued by the NMED in September 1991. The second assessment was designed to supplement the environmental monitoring data collected during the first assessment and to re-evaluate the potential extent and impact of groundwater contamination from the source waste oil tank. A series of new groundwater monitoring wells were installed and sampled between January 29, 1992, and May 17, 1993. The groundwater monitoring wells were installed to delineate both the horizontal and vertical extent of the contaminant plume. Data from this second assessment has been used to further define the probable nature and extent of contamination at the Person Generating Station site (e.g., METRIC, 1993).

1.2.1.2 Site Geology and Topography

The Person Generating Station site lies in the Rio Grande Basin, which has a general north-south alignment and is bordered on the east and west by upfaulted blocks. The basin is generally filled with unconsolidated alluvial material consisting of silt, clay, sand, and gravel. The upper geologic unit (Quaternary alluvium) is as much as 120 feet thick (Bjorklund and Maxwell, 1961; METRIC, 1993). The vadose zone is approximately 110 feet thick (METRIC, 1993). The upper geologic unit contacts the underlying unit between 80 and 120 feet below the land surface. The underlying geologic unit (Tertiary Santa Fe Group), which is at least 9000 feet thick in this area, is composed of unconsolidated and semiconsolidated sands, gravels, silts, and clays. Further detail on site geology can be found in earlier reports (e.g., Bjorklund and Maxwell, 1961; Lambert, 1968; Kelley, 1977; METRIC, 1993).

The regional topography of the Rio Grande Basin is marked by the Rio Grande River, which flows perennially north to south, approximately bisecting the alluvial valley and creating an alluvial floodplain to the east and west. Tijeras Arroyo, which has eroded through the land surface to the southeast of the Person Generating Station.

site, opens south onto the Rio Grande floodplain and flows to the southwest. The land surface in the vicinity of the Person Generating Station site slopes from 5 to 40 percent to the west. Local landform features are dissected terraces and alluvial fans. Within the Albuquerque Basin, elevations range from about 7300 feet in the mountains to the east of the Person Generating Station site to about 4900 feet at the Rio Grande to the west.

1.2.1.3 Site Hydrogeology

The principal aquifer of the Rio Grande Basin is the basin-fill Quaternary alluvium and Tertiary Santa Fe Formation sediments. The high capacity wells in the area typically tap the coarser grained river facies. The river facies extend from around 300 feet to around 1000 feet below the water table. There is continuous recharge of groundwater to and discharge of groundwater from the aquifer throughout the area. Recharge of the aquifer is the result of several processes, including infiltration of surface water from losing sources, such as the Rio Grande River and Tijeras Arroyo, direct recharge of precipitation, infiltration of excess irrigation water, and groundwater inflow from adjacent bedrock units and upstream basins.

Recent studies suggest that a hydrologic boundary between the recent alluvium deposit and the Santa Fe Group may not exist over a large area of the Rio Grande Basin (METRIC, 1993, and references therein). Groundwater circulates between the coarse unconsolidated beds of these units, driven toward discharge areas by higher hydraulic heads in the recharge areas. Units that are less permeable because they are fine-grained, cemented, or consolidated slow and/or deflect groundwater movement. The continuity of fine-grained units in the subsurface is unknown. The grain size may be coarser laterally and thus more permeable. Bjorklund and Maxwell (1961) reported an average hydraulic conductivity of 45 feet/day and a maximum of 112 feet/day for the Santa Fe Formation; the hydraulic conductivity of the river facies was expected to be in the upper end of this given range.

Earlier site assessments indicated that the groundwater generally flows southward within the Rio Grande Basin. However, in the local vicinity of the Person Generating Station site, particularly underlying the source area, the upper zone of groundwater flows about 82° east of south at a gradient of 0.43 percent. The groundwater gradient flattens to the east of Interstate 25 (METRIC, 1993). Potentiometric measurements also suggest the presence of a second flow zone (i.e., 25 to 35 feet below the water table) in which groundwater flows 83° east of south at a gradient of 0.48 percent.

Most recent groundwater monitoring data suggest a continual, general lowering of the water table under the Person Generating Station site at about a rate of 1 foot/year. This lowering may be caused by increased municipal pumping and decreased agricultural irrigation in the area.

1.2.2 Nature and Extent of Contamination

As discussed previously, PNM has documented the presence of several VOCs in the subsurface. The principal contaminants identified during monitoring activities are 1,1,1-TCA, PCE, and 1,1-dichloroethene (DCE). It appears that storage of rinse waters containing the Dowclene PC product, which contained significant concentrations

of two of the contaminants of concern (1,1,1-TCA and PCE), in the below-grade waste oil tank was the primary source of these contaminants. Historical documentation and site personnel interviews suggested that the concentration of these contaminants ultimately pumped to the waste oil tank may have been significantly decreased since a portion of these VOCs may have volatilized during maintenance operations (METRIC, 1993). Mixing the volatile products with large volumes of warm steam cleaning waste water may have enhanced the volatilization of contaminants of concern from the liquid waste transferred to the waste oil tank.

The presence of 1,1-DCE, which is not known to be a component of any of the liquid wastes introduced into the source waste oil tank, is likely attributable to the transformation of 1,1,1-TCA via hydrolysis. Although the reaction times of abiotic hydrolysis are relatively long (e.g., half-life of between 6 months to 1.5 years), these transformation reactions probably initiated in the source waste oil tank and continued during downward migration into and within the aquifer (Howard et al., 1991; Haag and Mill, 1988). Decreasing levels of 1,1,1-TCA and increasing levels of 1,1-DCE in the groundwater plume seem to support this transformation.

1.2.2.1 Existing Soil Contamination

The initial extent of soil contamination at the Person Generating Station site was originally reported in 1984 (Geoscience Consultants, Ltd., 1984). These and other site assessments documented high concentrations of VOCs in the upper 65 feet of the 110 feet thick vadose zone between the previous location of the tank bottom and the water table. There appears to have been downward migration of the VOCs through the unsaturated zone under the influence of gravity, most likely under unsaturated conditions. Laboratory analysis of soil borehole samples focused on PCE. Sample results show that the bulk of contaminated soil (delineated by soil concentrations of 1 ppm PCE or more) extends downward approximately 70 feet, is approximately 30 feet in diameter, and affects about 60,000 cubic feet of soil. The highest concentration of PCE measured in a soil sample was 2127 ppm at a depth of 15 feet from a borehole located in the center of the source waste tank area. However, data showed very low concentrations of VOCs in soil at a depth of 70 feet to the water table, which was located approximately 110 feet below ground level.

1.2.2.2 Existing Groundwater Contamination

The purpose of completed and ongoing groundwater assessments at the Person Generating Station site is to determine the horizontal and vertical extent and rate of movement of the VOC plume. Significant information on geological and hydrogeological characteristics, groundwater flow patterns, sample chemical analyses, and soil gas survey data have been compiled in the last ten years since the discovery of the leaking waste oil tank in October 1983. From this information, a summary description of the shallow VOC groundwater plume can be developed.

The horizontal groundwater plume boundary is defined in the Technical Schedule of the CAD as that location where contaminants of concern are at or above the target MCLs for each respective contaminant. The target MCL for 1,1,1-TCA is 60 ppb, 5 ppb for 1,1-DCE, and 5 ppb for PCE, per guidance from both the New Mexico Water Quality Control Commission and the U.S. Environmental Protection Agency. To

simplify analysis, the concentration value selected to operationally define the horizontal groundwater plume boundary was 5 ppb for each of the three contaminants of concern (i.e., PCE, 1,1,1-TCA, and 1,1-DCE). Using groundwater monitoring well data to develop plume concentration contour maps for the upper flow zone for each of the contaminants of concern, the total estimated areal extent of the VOC plume is currently 36 acres (METRIC, 1993). PCE and 1,1-DCE plumes extend downgradient to the east about 2400 feet from the source waste oil tank area. The 1,1,1-TCA plume extends about 1200 feet downgradient from the source waste oil tank area. The smaller horizontal extent of the 1,1,1-TCA plume may be attributed to a less concentrated source and/or effective *in situ* removal mechanisms in comparison to the other two contaminants of concern (e.g., due to pre-disposal evaporation during use and/or degradation to DCE via hydrolysis). Figure 1.2 illustrates the areal extent of the existing PCE plume, which is representative of the entire area affected by all three contaminants of concern (METRIC, 1993).

Monitoring well data from the last five years suggest that the plume may have reached its maximum areal extent and may actually be shrinking (METRIC, 1993). Additional data is necessary to verify the dominant role of natural fate and transport processes in controlling the horizontal migration of the shallow groundwater plume at the Person Generating Station site. It is conceivable that the horizontal extent of the shallow VOC groundwater plume may be largely attenuated due to the general lowering of the water table (at a rate of 1 foot/year) in the area. VOCs that have been adsorbed or absorbed onto immobile soil and organic material in the aquifer may be left stranded above the water table and unavailable to governing mass transport mechanisms (e.g., advective groundwater flow). It has been suggested that a portion of these VOCs may ultimately be transported upward through the unsaturated zone by diffusion in the soil gas and slowly discharged into the atmosphere (METRIC, 1993). Other subsurface characteristics (e.g., low local gradients, heterogeneous permeabilities) may limit the ability of the shallow aquifer to transport VOCs over large distances.

The vertical extent of the VOC groundwater plume is defined as the point at which the volatile contaminant levels are at or below the Method Detection Limits (MDL) values. The MDL level for the contaminants of concern at the Person Generating Station site is 0.2 ppb for Method 8010. Two separate sections can be used to describe vertical plume boundaries for each of the contaminants of concern at this site. The first vertical section is parallel to the axis of the plume; the second section is perpendicular to the axis of the plume. Using this operational definition, data indicate that the contaminants have not migrated beyond the upper flow zone, which is defined as 0 to 20 feet below the water table. Thus, the vertical extent of shallow groundwater contamination under the Person Generating Station site is about 20 feet below the water table.

Previous studies have demonstrated little potential for DNAPLs to exist in the aquifer underlying the Person Generating Station site (METRIC, 1993). The maximum concentrations ever detected in the groundwater are on the order of a few parts per million, which is less than 1 percent of the solubility of these chlorinated organics. The maximum concentration of a contaminant of concern ever measured in the shallow groundwater was 6.53 ppm (1,1,1-TCA). The aquifer in the vicinity of the source

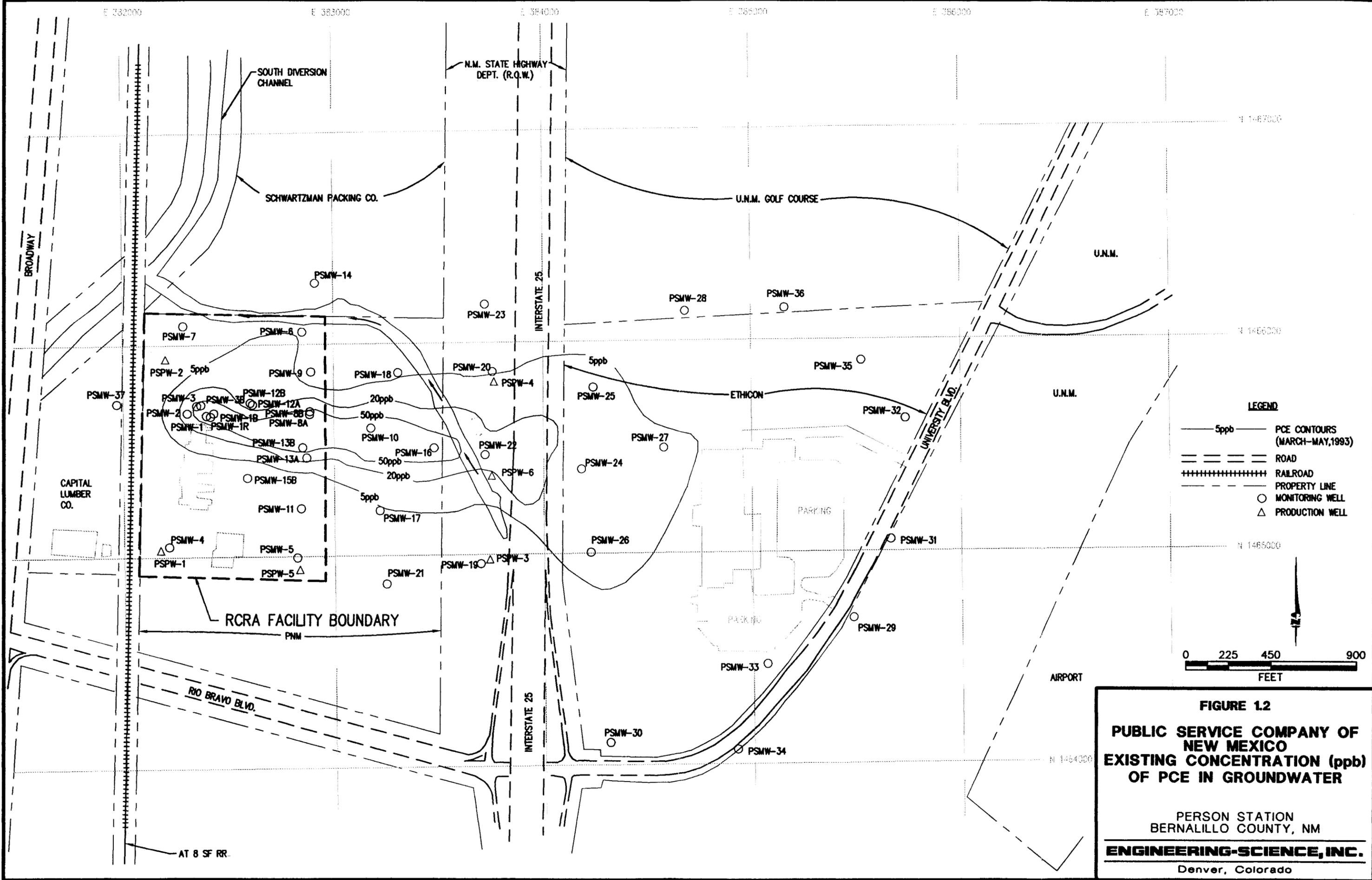


FIGURE 12
PUBLIC SERVICE COMPANY OF
NEW MEXICO
EXISTING CONCENTRATION (ppb)
OF PCE IN GROUNDWATER

PERSON STATION
 BERNALILLO COUNTY, NM

ENGINEERING-SCIENCE, INC.
 Denver, Colorado

waste oil tank has exhibited at least an order of magnitude decrease in contaminant concentrations since the tank was removed from service in 1983. This is also strong evidence that no DNAPL source exists beneath the waste tank location. Most recent data indicate a much lower maximum concentration of contaminants of concern in the groundwater (e.g., currently, 78 ppb of 1,1,1-TCA and a maximum VOC concentration of 670 ppb PCE) (METRIC, 1993). Without the continual addition of water to the waste tank, no significant mechanism for vertical transport through the vadose zone has existed since 1983.

SECTION 2

IDENTIFICATION AND DEVELOPMENT OF REMEDIAL TECHNOLOGY ALTERNATIVES

Available treatment technologies have been classified for potential use at the Person Generating Station site based on known contaminant properties, soil properties, existing site conditions, and estimates for remediation times. Brief descriptions of each technology are presented. Each technology was evaluated using the criteria specified in OSWER Directive 9902.3. In summary, this criteria focuses on short- and long-term effectiveness in protecting human health and the environment, reduction in contaminant toxicity, mobility, and/or volume, technical and administrative implementability, and relative cost to install, operate, and maintain the system. Those technologies identified as most promising after this initial evaluation were then combined into a conceptual remedial approach.

2.1 TECHNOLOGIES FOR *IN SITU* CONTAMINANT CONTAINMENT/DESTRUCTION

2.1.1 Hydraulic Containment

Hydraulic containment requires the installation of a line of wells at the downgradient edge of the plume to pump contaminated groundwater at such a rate that the migration of the contamination plume is halted. Hydraulic containment can be successfully applied in either shallow or deep aquifers. It differs from a pump-and-treat system because the objective is only to intercept the plume, not remove large quantities of groundwater throughout the entire plume area. The pumped groundwater can either be treated and recycled through the contaminated regions of the site (soil washing), or discharged to surface waters or a sanitary sewer.

Establishing a hydraulic containment system requires a thorough understanding of the underlying hydrogeology. Computer modeling of the area can demonstrate the effectiveness of hydraulic containment and give an indication of the number of wells required to contain a contamination plume.

2.1.2 Semipermeable Barriers

Semipermeable barrier applications require the installation of a semipermeable barrier wall that treats the contaminated ground water *in situ* as it passes through the barrier wall. One potential application involves the installation of an activated carbon zone to actively adsorb the contaminants. Activated carbon applications are generally used in shallow aquifer systems. In order to construct a uniform and homogeneous

barrier wall, the construction sequence usually involves driving sheet piling into the confining layer, dewatering the barrier area, replacing the native soil with activated carbon, and removing the sheet piling.

Iron filings also have been used in semipermeable barrier applications as a method of chemically destroying halogenated organic compounds *in situ* (O'Hannesin, and Gillham 1993). However, iron filing barriers are not a proven treatment process and require further research prior to full-scale applications. Furthermore, the probability of incomplete reactions and forming toxic by-products is also increased in uncontrolled subsurface applications.

2.1.3 Enhanced Biodegradation

In situ biodegradation is a technology which uses acclimated, indigenous bacteria to degrade the contaminants of concern within the aquifer. Although this technology is widely applied to fuel hydrocarbons, to date its application to chlorinated hydrocarbons has been limited to research.

Enhanced biodegradation of chlorinated compounds with aerobic bacteria involves the addition of a cosubstrate, such as methane, and large quantities of oxygen to enhance the natural biodegradation of the chlorinated organic compounds. A pilot test, which attempted to remediate trichloroethylene (TCE) *in situ* using this technique, produced inconclusive results due to the lack of control and mixing within the aquifer (Semprini, 1987). The application of this technology for groundwater remediation is further complicated by the fact that PCE is not degraded by aerobic bacteria. While PCE can be degraded by anaerobic bacteria, this process has only been successfully documented using laboratory bioreactors. Anaerobic conditions would be very difficult to create *in situ* without adding large quantities of a cosubstrate oxygen consumer such as acetate (Engineering-Science, Inc., 1991).

2.1.4 Natural Attenuation

Natural attenuation is a process where dispersion, adsorption, chemical hydrolysis, and biodegradation gradually reduce the concentrations of chemicals present in the soil and groundwater. Contaminants are often dispersed in high conductivity aquifers or adsorbed on soil particles and removed from the groundwater plume. At the Person Generating Station site, evidence suggests that adsorption processes may be occurring and that contaminants adsorbed to soils are probably removed from the groundwater as the water levels continue to drop in this area at a rate of one foot per year. It does not appear that a significant level of dispersion is occurring, however, possibly due to several limiting characteristics of the shallow aquifer (e.g., low local gradients, variable permeabilities).

Chemical hydrolysis is the direct reaction of dissolved compounds with water molecules. The hydrolysis of chlorinated compounds ultimately yields an alcohol or alkene (U.S. Environmental Protection Agency, 1989). The rate of hydrolysis is affected by temperature, solvent composition, catalysis, and pH (U.S. Environmental Protection Agency, 1985). PCE and 1,1-DCE do not readily degrade through hydrolysis under normal environmental conditions. However, 1,1,1-TCA will hydrolyze to 1,1-DCE, with an estimated half-life of six months to 1.5 years (Haag and

Mill, 1988; Camp, Dresser, & McKee, Inc., 1990). No additional hydrolysis or biodegradation by-products, such as vinyl chloride, have been observed at this site.

Both 1,1,1-TCA and 1,1-DCE can be aerobically degraded under natural conditions, however, this process is expected to be very slow since it depends on indigenous microorganisms and natural nutrients which are not abundant in deep sandy aquifers. PCE is not degraded under the aerobic conditions which should prevail in this aquifer.

2.2 TECHNOLOGIES FOR CONTAMINANT REMOVAL

2.2.1 Pumping

Pumping is the direct removal of contaminated groundwater using a well, or system of wells, in the plume area. Pumping rates and well locations will vary depending on site hydrogeology. Submersible pumps are generally used to remove deep groundwater.

Groundwater pumping is an effective method of removing dissolved contaminants, but is less effective for chemicals which are strongly adsorbed to soil particles. Another limitation of groundwater pumping systems is their inability to rapidly remove contaminants that are slowly dissolved from a pure non-aqueous phase. However, DNAPL contaminants are not expected at the Person Generating Station site.

2.2.2 Vapor Extraction

Vapor extraction is a cost effective technology using vertical or horizontal vent wells to rapidly extract and collect contaminated soil gas. Vapor extraction is particularly effective in sandy soils where higher volumes of soil vapor can be extracted through the use of a vacuum. Due to their relatively high vapor pressures, 1,1,1-TCA, 1,1-DCE and PCE have been effectively removed from soils utilizing this technology.

2.2.3 Air Sparging

Air sparging techniques have been used to treat VOC contaminated saturated soil and groundwater. Air sparging involves the installation of a number of sparging (injection) wells to inject high-pressure air into the saturated zone into the areas of contamination. The contaminants dissolved in the groundwater partition into the air phase and are transported into the soil vadose zone for collection by a vapor extraction system.

In order to avoid the horizontal displacement of contaminated vapor and contaminated groundwater, an air sparging treatment system must be carefully designed. Furthermore, potential problems with iron oxidation and aquifer plugging exist whenever air is introduced into groundwater.

2.3 TECHNOLOGIES FOR ABOVEGROUND TREATMENT

2.3.1 Air Stripping

Air stripping is a mass-transfer unit operation in which the volatile contaminants dissolved in the groundwater are transferred from the liquid phase to the vapor phase. Generally constituents with Henry's Law constants of greater than $0.003 \text{ atm}\cdot\text{m}^3/\text{mole}$ can be effectively removed by air stripping (Conway and Ross, 1980). All of the

contaminants of concern at this site have Henry's Law constants greater than 0.006 atm-m³/mole and could be effectively removed using air stripping technology. (U.S. Environmental Protection Agency, 1985).

Air stripping operations require a feed stream low in suspended solids. The feed stream also may require pH adjustment in order to reduce the precipitation of iron and manganese on the packing material. Low-profile, tray type air stripping units can reduce the impact of mineral fouling and are easier to maintain. In some cases, an activated carbon unit may be necessary to polish the air stripped liquid effluent in order to meet regulatory discharge requirements. Air stripper vapor phase effluents may require air emission controls in more contaminated groundwater treatment applications, but should not be a requirement at this site.

2.3.2 Activated Carbon

Activated carbon units can be used to remove chlorinated compounds from both liquid and/or vapor phase waste streams. Activated carbon units can be installed within a pumping system to directly treat the groundwater or can be used to remove organics from the vapors extracted from soil venting operations.

The process of removing contaminants through adsorption onto the activated carbon involves passing the waste stream through one or a series of packed-bed carbon vessels or exchangeable canisters. The effectiveness and longevity of the adsorption system will vary with the concentration of the waste stream and the capacity of the canisters.

Utilizing liquid-phase activated carbon units as the primary organic removal step is generally expensive and requires more frequent change out periods. "Spent" carbon units also can be listed as a hazardous waste requiring special handling and regeneration.

2.3.3 Biological Treatment

In biological treatment reactors, microorganisms metabolize organic compounds as their primary carbon source for growth and metabolism. In some cases, the organic compounds (such as chlorinated solvents) must be co-metabolized with other easy to degrade compounds such as methane. Biological treatment can occur under aerobic or anaerobic conditions. To date, full-scale applications have been limited to low flow rate reactors, and are often followed by activated carbon for polishing.

PCE is resistant to aerobic degradation, but can be anaerobically degraded. However, by-products can result from the partial degradation of some chlorinated compounds. PCE, if incompletely degraded under anaerobic conditions, may degrade to intermediates such as 1,1,1-TCE, 1,1-DCE, and vinyl chloride (Bourguin, 1989; Bower and McCarty, 1984; and Schraa et al., 1984).

2.3.4 Catalytic Oxidation

Catalytic oxidation units, also known as catalytic incinerators, are used to treat waste air streams contaminated with VOCs collected from vapor extraction or air stripping operations. Catalytic oxidation units operate at lower temperatures than thermal incinerator units, but still maintain high destruction efficiencies. Consequently, less fuel is required to operate the catalytic oxidation unit.

A catalytic oxidation unit destroys the VOCs in the air stream by contacting the air with a fluidized bed of catalyst granules that are maintained at a controlled temperature (generally greater than 700° Fahrenheit). Natural gas is used to heat the catalyst and preheat the air stream. TCE was successfully removed from an air stripper effluent, at efficiencies of 95 to 99 percent, in a unit located at Wurtsmith Air Force Base, Oscoda, Michigan (Air Force Engineering and Services Center, 1992).

SECTION 3

SCREENING AND INITIAL EVALUATION OF REMEDIAL TECHNOLOGIES

Many of the remedial technologies described above may be appropriate for use at the Person Generating Station site. To identify those technologies that may be most appropriate for this application, each technology must be evaluated. The criteria used to evaluate remedial technologies were adapted from those recommended by the U.S. Environmental Protection Agency for selecting remedies for Superfund sites (OSWER Directive 9902.3). The criteria used included (1) short- and long-term performance in protecting human health and the environment, (2) ability to reduce contaminant toxicity, mobility, and/or volume, (3) technical and administrative implementability, and (4) relative cost. This evaluation seeks to identify only the most promising technologies. Table 3.1 summarizes the results of this evaluation process.

3.1 HYDRAULIC CONTAINMENT

Hydraulic containment is rejected because it will not rapidly reduce contaminant concentrations in the 36-acre plume. The effectiveness of a hydraulic containment system ultimately depends upon the number and alignment of the installed wells, the accuracy of the initial hydrologic investigation, and the ability of the system to adjust to changing groundwater conditions. Historically, hydraulic containment systems have been effective in shallow, contaminated aquifers. However, over larger areas, hydraulic containment is not effective in rapidly reducing source area contaminants. Although the majority of the contaminants at the Person Generating Station site are within a 100-foot radius of the former source waste oil tank location, containment alone would require far more than 20 years to reduce overall contaminant concentrations below target MCLs.

3.2 SEMIPERMEABLE BARRIERS

Semipermeable barrier *in situ* treatment technology is rejected due to the depth of the unconfined aquifer and the inability to ensure proper treatment of the site contaminants. The depth to groundwater at the Person Generating Station site is over 200 feet below the ground surface in some locations. The construction of a semipermeable barrier would be very expensive since it would require deep slurry trench construction techniques. Furthermore, at these depths, construction quality controls would be very difficult to confirm.

TABLE 3.1

SUMMARY OF REMEDIAL TECHNOLOGY SCREENING

General	Technology	Decision	Performance	Reliability	Implementability
<i>In situ</i> Contaminant Containment/ Destruction	Hydraulic Containment	Reject	Will not rapidly reduce VOC plume; require more than 20 years	Not reliable over large areas	Effectiveness depends on well system alignment
	Semi-permeable barriers	Reject	Unproven in deep aquifers	Cannot ensure proper treatment	High cost, high material requirements
	Enhanced biodegradation	Reject	Difficult to degrade PCE	Unproven full-scale	Difficult to insure uniformity
	Natural Attenuation	Reject	Can not rapidly remove source of VOC plume	Slow but will occur	Unacceptably long treatment time
Contaminant Removal	Pumping	Retain	Quickly removes high VOC concentrations; requires concurrent treatment technology	Proven; minimal risk of equipment failure; require long-term operation/ maintenance	May not reach MCLs
	Vapor Extraction	Retain	Effectively and quickly removes VOCs from vadose zone	Proven; minimal risk of equipment failure	May require vapor treatment

3-2

TABLE 3.1**SUMMARY OF REMEDIAL TECHNOLOGY SCREENING (Continued)**

General	Technology	Decision	Performance	Reliability	Implementability
	Air Sparging	Reject	Can not predict effectiveness given size of VOC plume	Unable to predict range of influence; problem with short-circuiting	High cost; extensive equipment requirements
Above-ground Treatment	Air Stripping	Retain	Effectively removes 99% VOCs from extracted water; may need pretreatment	Proven; moderate maintenance required	No constraints identified
	Activated carbon	Reject (pending Phase I pilot tests)	Effectively removes VOCs from extracted water and vapor; also effective as polishing step	Proven	High cost compared to air stripper technologies; generates waste
	Biological treatment	Reject	Not effective under high flow conditions	Cannot ensure complete treatment	Not proven for full-scale
	Catalytic oxidation	Reject (pending Phase I pilot tests)	Effective on high VOC concentrations from vapor extraction	Proven	High cost; extensive maintenance requirements

3.3 ENHANCED BIODEGRADATION

In situ enhanced biodegradation is rejected as a treatment alternative since PCE is not readily degraded by aerobic bacteria. Furthermore, the large area requiring treatment would make enhanced biodegradation difficult to uniformly apply.

3.4 NATURAL ATTENUATION

Natural attenuation is rejected as a "primary" treatment alternative since the source and migration of the contaminant plume would not be immediately controlled. Although the continuing drop in the water table in this area would likely enhance removal of contaminants from groundwater and adsorption to soils above the receding groundwater, these processes alone will not reduce contamination below target MCLs. Some biological and chemical degradation of these contaminants will continue to occur, however.

PCE will not degrade naturally in aerobic environments and will be persistent across the site. 1,1-DCE may slowly degrade across the site, while 1,1,1-TCA may degrade through both biodegradation and hydrolysis across the site.

Natural removal and dispersion processes may be used at the site for areas where the contamination levels are approaching MCLs. However, natural attenuation will only be effective at this site when used in conjunction with other source removal technologies.

3.5 PUMPING

3.5.1 Performance

3.5.1.1 Effectiveness

Direct pumping of contaminated groundwater could be effective in removing PCE, 1,1-DCE, and 1,1,1-TCA from the aquifer. Direct pumping would be particularly effective in removing high level contamination from the source area at the Person Generating Station site. No DNAPLs have been observed during site investigations; groundwater concentrations of these contaminants are orders of magnitude below the solubility limits for these constituents indicating no DNAPL source is likely to be present.

The sandy soils of the underlying aquifer should generally enhance the effectiveness of a pump-and-treat technology. Sandy aquifer material does not strongly adsorb organic contaminants and is more permeable, allowing more rapid removal of the contaminants from the aquifer. A pump-and-treat process could also use some of the existing 4-inch wells at the site.

Direct pumping would require the above-ground treatment of the groundwater and will not remove vadose zone contamination. Therefore, this technology would require the concurrent use of another technology, such as soil vapor extraction, to reduce the contaminant levels in the source area vadose zone.

3.5.1.2 Useful Life

Pumping systems can be used for extended periods of time. The useful life of the system will generally depend on the quality of the equipment used to pump the groundwater and the use of properly constructed wells. Due to continuous use, pumps may need to be replaced every two to three years. However, with proper maintenance of wells and equipment, the performance of the system should not significantly deteriorate over time.

3.5.2 Reliability

3.5.2.1 Operation and Maintenance Requirements

Operation and maintenance requirements of a long-term pumping operation could be considerable. Operation activities would include flow adjustments, well screen cleaning, and pump replacements. Maintenance activities would be conducted according to manufacturer's literature on equipment maintenance. Further, appropriate discharge/recharge areas and/or beneficial uses for removed groundwater would have to be identified. All resources necessary to perform the operation and maintenance can be found in the local area.

3.5.2.2 Demonstrated and Expected Reliability

Groundwater pumping technologies require long-term operation and maintenance. However, pump-and-treat methods have been successfully applied to groundwater remediation projects to remove high levels of dissolved contaminants when DNAPLs are not present. Examples of the application of pump-and-treat technologies for chlorinated organics are provided in Appendix C. The risk and effect of a temporary system failure would be minimal since a few feet of groundwater advance would pose little risk to surrounding populations. All of the equipment can be serviced by companies in the local area.

3.5.3 Implementability

3.5.3.1 Constructability

The constructability of a pumping system can be limited by geological and hydrological characteristics of a site and the geographic location of a site. Since the Person Generating Station site primarily consists of sandy soils within an unconfined aquifer, no subsurface constructability constraints should exist. Steep terrain and an interstate highway on the eastern edge of the plume could create some challenges to well and piping construction. Basic well construction principles would be used to construct the wells. Previous drilling operations at the site have been successful.

It is estimated that a system of 4-6 wells located along the plume centerline would be necessary to contain and pump the contaminated groundwater with total VOCs currently exceeding 10 ppb.

3.5.3.2 Schedule Considerations

Since standard methods would be used in constructing a pumping well network, no time limitations should influence the implementation of this method as a corrective

measure. A construction time of approximately 60 days is anticipated for the well network.

Beneficial results should be seen by the rapid reduction of contamination levels at the source area. Pulsed pumping could be used to reduce the total quantity of groundwater pumped, hence, reducing the cost of above-ground treatment. Optimum selection of pumping locations would also reduce the time necessary to treat contaminated groundwater.

3.5.3.3 Safety

The construction of the system would involve standard safety methods used in the installation of groundwater wells. Well installation is a common practice that follows the guidelines set forth by OSHA. No unique construction techniques would be required. Adjacent areas would not see any additional safety threats from the installation and operation of a pumping network. Furthermore, a system shut down would not create a hazardous situation.

3.6 VAPOR EXTRACTION

3.6.1 Performance

3.6.1.1 Effectiveness

Vapor extraction is a low cost, proven technology that efficiently removes VOCs from the soil vadose zone. Based on ES experience, the removal of the target VOCs will be rapid, with more than 90 percent (by weight) of the removal occurring in the first six months of operation. A greater than 95 percent removal rate of source contaminants could be expected after the first year of operation. Examples of the successful application of soil vapor extraction for the removal of chlorinated organics are also provided in Appendix C.

The vapor pressures of PCE, 1,1-DCE, and 1,1,1-TCA support the use of vapor extraction for the removal of these constituents from the soil media. Offgas treatment, such as activated carbon, may be required in order to conform to air emission standards or to ensure safe conditions for site workers.

3.6.1.2 Useful Life

Vapor extraction systems can be used for extended periods of time. Vapor extraction would be used to treat the contaminated vadose zone near the source area and, therefore, may not require an extended operational period. Therefore, the useful life of an installed vapor extraction system should exceed the required operational life of the system. However, if extended operations are deemed necessary, vacuum blower or motors may need to be repaired or replaced every 3-4 years.

3.6.2 Reliability

3.6.2.1 Operation and Maintenance Requirements

Operation and maintenance of a short-term vapor extraction system should be minimal. Operational activities would include flow adjustments, sample collection for performance monitoring, and possible transfer of condensate to the groundwater

treatment system. Maintenance activities would be conducted according to manufacturer literature on equipment maintenance. All resources necessary to perform operation and maintenance can be found in the local area.

3.6.2.2 Demonstrated and Expected Reliability

Vapor extraction technology has been successfully demonstrated for a number of years under conditions similar to those encountered at the Person Generating Station site. A vapor extraction system, consisting of one or two wells, should be able to operate efficiently under the sandy soil and anticipated contaminant conditions. Vapor extraction flow rates can be adjusted to optimize the treatment process.

3.6.3 Implementability

3.6.3.1 Constructability

Vapor extraction wells are constructed using standard methods developed by the remediation industry. Therefore, since previous drilling operations at the site have been successful in the sandy soil, it is assumed that vapor extraction well installation also should be successful.

During the construction of the wells, it is important to correctly seal the top of the well with bentonite to prevent short circuiting. A detailed construction process would be outlined prior to the application of this technology.

3.6.3.2. Schedule Considerations

Since standard well construction methods and equipment would be utilized in constructing a vapor extraction system, no special scheduling will be required in the implementation of this remediation method. Beneficial results should be instantaneous. Vadose zone contamination levels should be rapidly reduced.

3.6.3.3 Safety

The construction of a vapor extraction system would involve the standard safety methods used in the installation of groundwater wells. Well installation is a common practice that follows the guidelines set forth by OSHA. No unique construction techniques or equipment would be required.

The local area would not see any additional safety threats from the installation and operation of such a system. Emissions of chlorinated solvents to the atmosphere will occur but long-term concentrations will be low and the relative isolation of the site should not lead to health risks. Atmospheric dispersion models could be used to verify that VOCs will not pose a health risk to site workers or nearby populations. During the initial months of operation, activated carbon may be required to meet air discharge standards.

3.7 AIR SPARGING

Air sparging is rejected as a treatment technology. Though air sparging can be effective in sandy soils, it is difficult to predict the effective radius of influence for each well. Recent research has shown problems exist with air sparging uniformity in the saturated zone. Significant short-circuiting and channeling has been observed in

controlled full-scale research projects (Johnson, 1993). A reasonable estimate of a full-scale radius of influence would be 20 feet per sparge point. Based on a plume area that is 1800 feet by 500 feet (with total VOC levels exceeding 20 ppb), over 800 sparge points would be required.

The use of both air sparging and vapor extraction wells to remove VOCs from the vadose zone also would increase the cost of applying this technology. Hence, due to the estimated cost to install an air sparging system to treat the contamination plume, a concern of nonuniform treatment, and the possibility of contributing to contaminant migration, air sparging is rejected as a treatment alternative.

3.8 AIR STRIPPING

3.8.1 Performance

3.8.1.1 Effectiveness

Air stripping technologies are a proven process that have been used to remove VOCs from groundwater. A 95-99 percent removal rate for PCE, 1,1-DCE, and 1,1,1-TCA would be expected using air stripper technology (Engineering-Science, Inc., 1991). Air emissions are expected to be minimal due to the low levels of groundwater contamination. No site characteristics should impede the performance of an air stripping operation.

3.8.1.2 Useful life

Air stripping technology could be successfully applied to the site for a number of years. However, continuous operation may require that blower motors be replaced or repaired every few years. The use of tray type air stripping units could significantly simplify the removal of mineral deposits and increase system life.

3.8.2 Reliability

3.8.2.1 Operation and Maintenance Requirements

Start-up and shut-down of air stripping systems is relatively quick and simple. Operation activities would include air and water flow adjustments and influent and effluent monitoring. Maintenance activities may include blower replacement and general manufacturer recommended equipment maintenance. Packing material may require periodic cleaning. Temporary shut down for cleaning would have no adverse impacts on long-term remediation. All resources necessary to perform operation and maintenance procedures can be found in the local area.

3.8.2.2 Demonstrated and Expected Reliability

Air stripping technologies are mechanically simple and have demonstrated long-term reliability if properly maintained.

3.8.3 Implementability

3.8.3.1 Constructability

No apparent constructability factors should inhibit the utilization of air stripping technology. A proven, preengineered, skid-mounted system would be recommended for the Person Generating Station site.

3.8.3.2 Schedule Considerations

The time involved to acquire and install the air stripping unit should be less than the time required to install pumping wells across the site. The system should be located approximately 150 feet north of the power plant for easy access.

3.8.3.3 Safety

Adjacent areas would not see any additional safety threats from the installation and operation of such a system. Air emissions would be minimal due to the very low concentrations of VOCs in the extracted groundwater. The system should be equipped with an emergency shut down switch which shuts off pumping wells if the air stripper shuts down.

3.9 ACTIVATED CARBON

3.9.1 Performance

3.9.1.1 Effectiveness

Activated carbon is a well-developed technology which is widely used to treat contaminated groundwater and vapor waste streams. An activated carbon process should efficiently remove low levels of PCE, 1,1-DCE, and 1,1,1-TCA from liquid or vapor waste streams. However, as a primary treatment process, activated carbon treatment could be considerably more expensive than air stripping technologies. Furthermore, the "spent" activated carbon units could be considered a hazardous waste and require proper handling, labeling, and transportation to the manufacturer for regeneration.

The method of utilizing activated carbon as the "primary" contaminant removal process for extracted groundwater is rejected due to the significant cost associated with operating and maintaining such a system. However, carbon alternatives are being retained for use in treating soil vapor extraction emissions if regulatory requirements require this treatment.

3.9.1.2 Useful Life

The service life of a carbon polishing operation would vary depending on the effluent concentrations of the contaminants and the capacity of the carbon units. In a polishing operation, change out periods would be far less frequent than that expected in a primary treatment operation. Resource availability should be acceptable, since carbon unit replacement would be coordinated with the manufacturer before contaminant "breakthrough" occurred.

3.9.2 Reliability

3.9.2.1 Operation and Maintenance Requirements

Operational considerations include possible performance monitoring of the carbon unit(s) to determine breakthrough times. Predetermined change out periods, with an acceptable safety factor, also could be applied to determine change out periods. Extra carbon units also may be stored onsite if necessary.

The required maintenance of a closed system activated carbon unit is minimal. However, proper labeling, storage, and shipment of the spent units, possibly as a hazardous waste, must be ensured.

3.9.2.2 Demonstrated and Expected Reliability

Activated carbon technologies have been successfully demonstrated under many contaminant conditions. The moderate levels of VOCs anticipated in extracted soil gas from the source area may require the use of activated carbon units in the early stages of soil vapor extraction activities. PCE, 1,1-DCE, and 1,1,1-TCA are all removed from liquid or vapor phases when passed through activated carbon units (Engineering-Science, Inc., 1991). Carbon unit fouling will not be a factor in vapor phase treatments.

3.9.3 Implementability

3.9.3.1 Constructability

Installation of the carbon units would involve the simple attachment of the units to the proper effluent ports. No significant startup delays are anticipated.

3.9.3.2 Schedule Considerations

The time needed to implement this technology would be less than 30 days. Beneficial results would be instantaneous.

3.9.3.3 Safety

Adjacent areas would not be exposed to any additional safety threats from the installation and operation of such a system. Spent carbon units may need to be handled as hazardous waste and properly transported to the supplier for regeneration.

3.10 BIOLOGICAL TREATMENT

A biological treatment system is rejected as a treatment technology due to the anticipated high flow rates. The necessary high flow rates at the Person Generating Station site will not provide adequate contact-times within a biological treatment unit. Furthermore, PCE cannot be aerobically degraded and would require an anaerobic treatment system with a long residence time. Therefore, the complexity of the system would lead to an inability to accurately estimate and control removal rates.

3.11 CATALYTIC OXIDATION

Catalytic oxidation is rejected as a treatment option due to the relatively low concentrations of contaminants anticipated in the vapor streams at the Person

Generating Station site and the high cost and maintenance requirements of this technology when compared to other technologies. Vapor concentrations from the soil vapor extraction unit will be determined during Phase I pilot tests. If those concentrations are higher than anticipated and require treatment, the catalytic oxidation option could be reconsidered.

SECTION 4

RECOMMENDED REMEDIAL APPROACH AND CONCEPTUAL DESIGN

ES recommends a two-phase approach to significantly improve the efficiency of full-scale remediation at the Person Generating Station site. Rather than attempt to design a full-scale remediation system based on the success of specific technologies at other sites, ES recommends a Phase I pilot study to determine the optimum application of technologies prior to full-scale design. Phase I would begin to remediate the soil and groundwater within a 100-foot radius of the former waste oil tank. The estimated time to completion for Phase I operations is three months. Information gathered during Phase I operations would then be used to optimize the design and operation of the treatment system for the rest of the contaminated plume area. Phase II operations would include continued source area remediation and be expanded to include remediation of contaminated groundwater downgradient of the former waste oil tank area.

4.1 OVERVIEW

No single treatment technology can be feasibly or economically applied to clean up both the soil and groundwater contamination at the Person Generating Station site. Therefore, in order to optimize the removal of vadose zone soil and groundwater contamination and reduce overall treatment times, the integration of several treatment technologies has been recommended.

The recommended Phase I treatment methods are divided according to media. The first methods are for the removal and treatment of the contaminated groundwater in the source area. Groundwater would be pumped using one new 4-inch well in the source area and one existing 4-inch monitoring well (PSMW-16). The groundwater would then be passed through an air stripper process to remove the VOCs. The treated groundwater could be possibly discharged to the stormwater drainage northwest of the site in accordance with applicable discharge permit standards. Phase I treatment also would involve treating the contaminated soil in the vicinity of the former waste oil tank. Soil vapor extraction methods would be used to extract contaminated soil gas and volatilize contaminants from the soil. If the levels of extracted VOCs do not constitute a health risk, the extracted soil gas could be discharged directly to the atmosphere. If offgas treatment is required, ES proposes the use of a vapor-phase carbon unit to capture volatile contaminants.

Phase II operations would utilize the site-specific information obtained from Phase I operations. Phase I operations will continue throughout Phase II. Phase II will consist of pumping and treating the contaminated groundwater plume that exceeds a total VOC concentration level of approximately 10 ppb with a goal of decreasing each contaminant concentration to less than 5 ppb at all points in the shallow aquifer. Groundwater would be pumped from a network of wells and sent to the central treatment system. The groundwater would then be passed through an air stripper process to remove the VOCs. The offgas from the air stripper would be released to the atmosphere. The clean groundwater would then be discharged to a nearby storm drainage channel under PNM's pending National Pollution Discharge Elimination System (NPDES) permit or used for irrigation.

The actual well spacing and number of wells for Phase II will be based on data collected during Phase I pumping operations and groundwater model predictions. Optimization methods, such as pulsed pumping, may also be integrated into the remediation system as more information is gained.

4.2 PHASE I CONCEPTUAL DESIGN

4.2.1 Soil Vapor Extraction

A 60- to 70-day soil vapor extraction test will be conducted in the soil contamination source area near the former waste tank. A single 4-inch PVC vapor extraction well/dewatering well (VEW/DW) will be constructed through the existing concrete cap with a screened interval from 10 feet below ground surface to approximately 145 feet. Figure 4.1 illustrates the proposed location of this dual-purpose well and construction details. By placing this soil vapor extraction well/dewatering well near the center of the spill, maximum recovery of volatile contaminants should occur. The special surface completion of this well will insure the integrity of the concrete cap.

During Phase I testing, a 10-HP test blower will be connected to the VEW/DW and used to extract approximately 100 scfm of soil gas from the source area. Initial soil vapor concentrations of PCE, 1,1,1-TCA and 1,1-DCE are expected to total over 1000 ppmv and activated carbon canisters are proposed to remove these high initial levels of vapor contamination prior to discharge to the atmosphere. Carbon canisters may not be required for long-term operations because vapor concentrations are expected to decrease rapidly. An air discharge permit will be required from Bernalillo County and the degree of vapor treatment will be determined by modeling of the impact of these VOCs on ambient air quality.

A schematic of the vapor extraction system is included as Figure 4.2. This test unit is a preengineered, trailer-mounted system owned by Engineering-Science, Inc. and will not require additional design or construction. Flow rates, contaminant concentrations, and blower temperature will be monitored at regular intervals to insure optimum extraction rates are maintained. Spent carbon canisters may require handling as hazardous waste and will be returned to the manufacturer for regeneration.

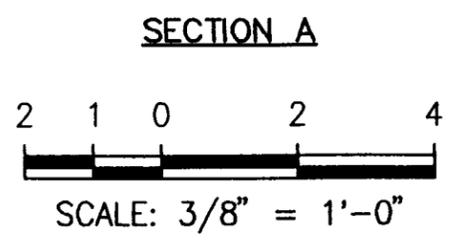
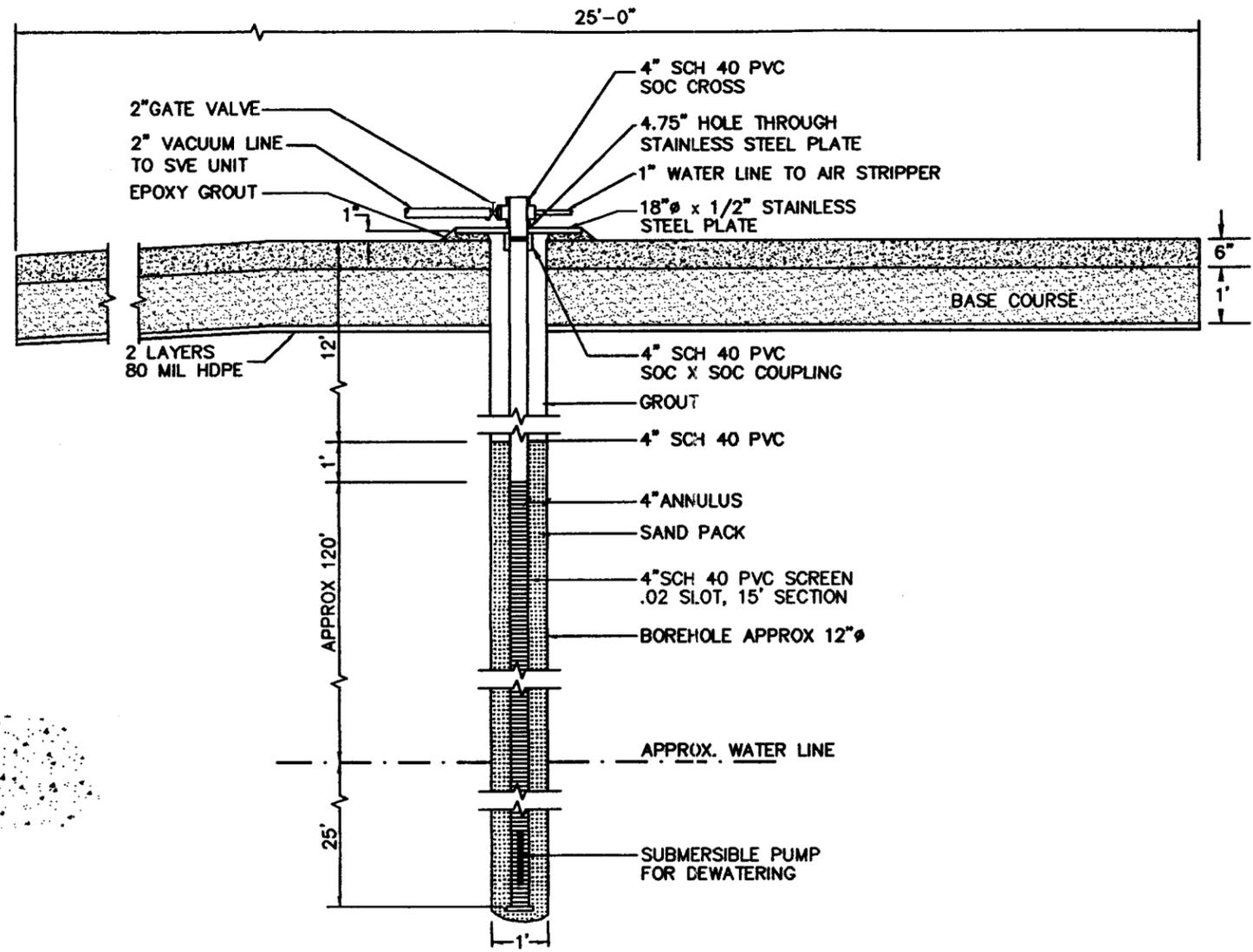
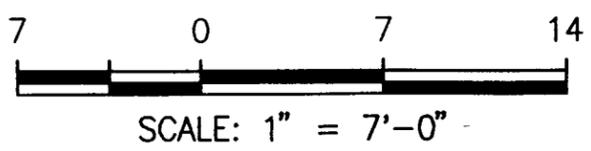
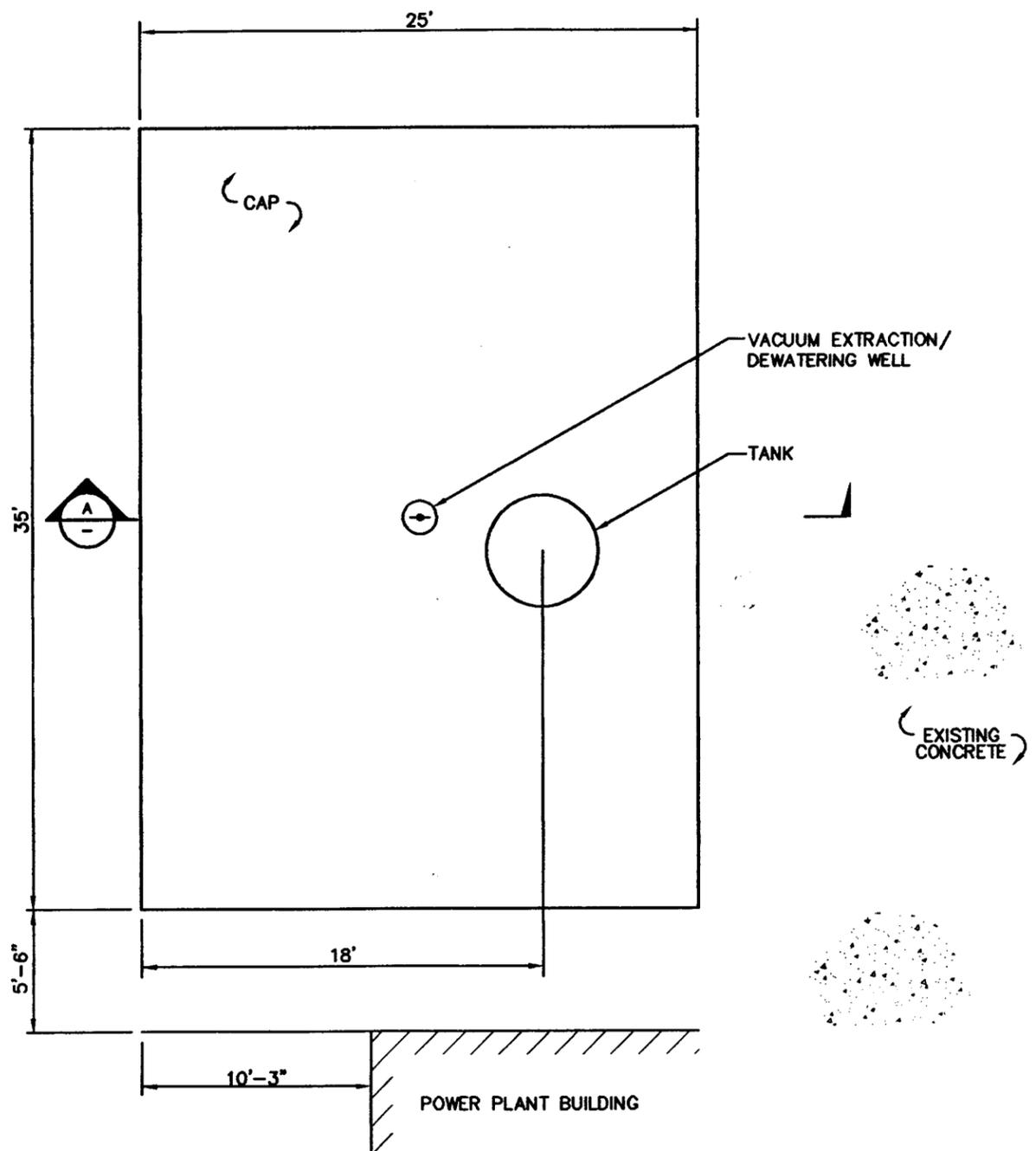
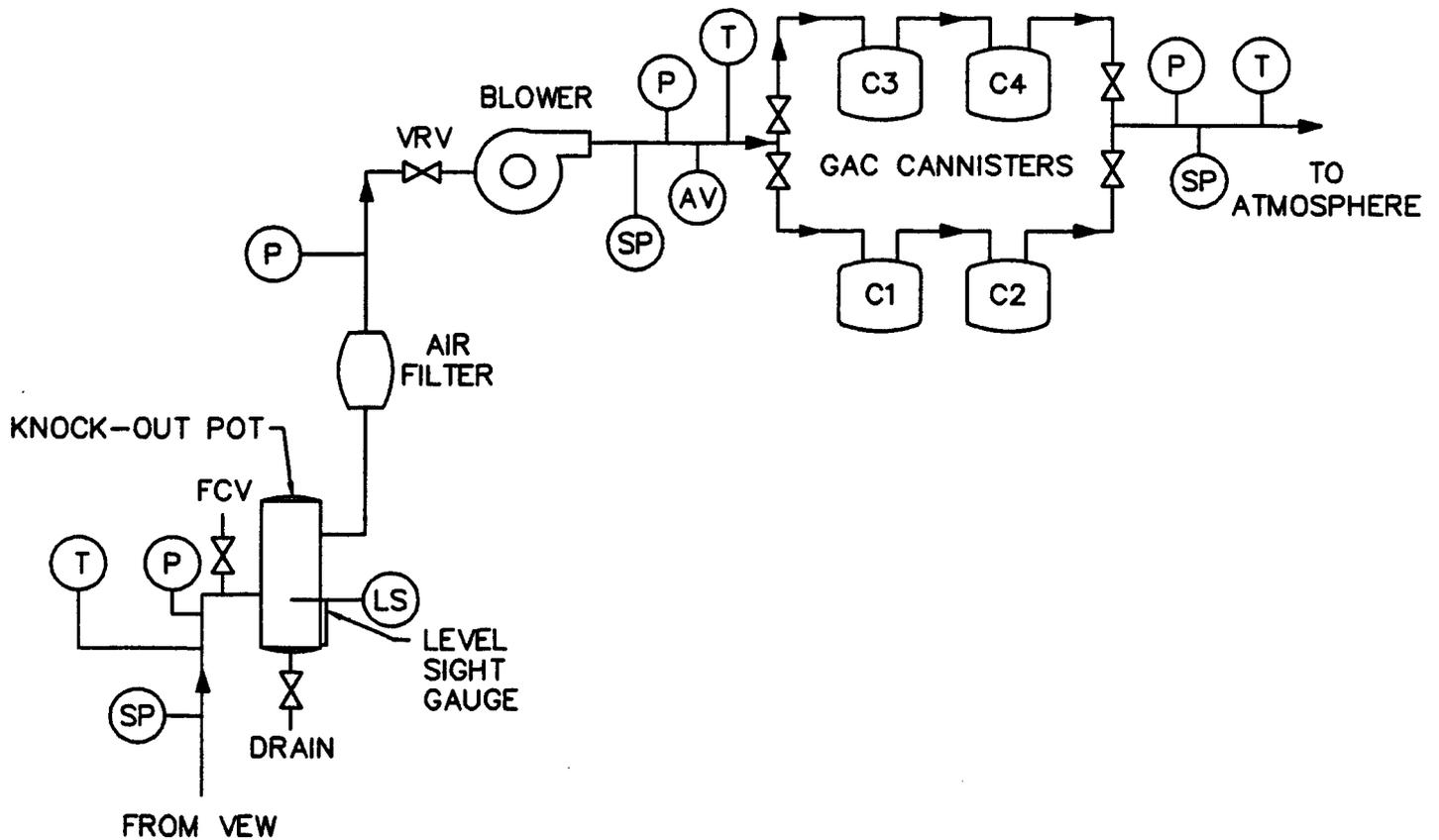


FIGURE 4.1
PHASE II
VAPOR EXTRACTION
DEWATERING WELL (VEW/DW)

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LEGEND

- (SP) SAMPLE POINT
- (AV) AIR VELOCITY PITOT TUBE
- (P) PRESSURE GAUGE
- (T) TEMPERATURE GAUGE
- (LS) LEVEL SWITCH
- FCV FLOW CONTROL VALVE
- VRV VACUUM RELIEF VALVE
- GAC GRANULATED ACTIVATED CARBON

FIGURE 4.2
PROCESS FLOW AND
INSTRUMENTATION DIAGRAM
SOIL VAPOR EXTRACTION
SYSTEM WITH CARBON
TREATMENT
PERSON GENERATING SITE

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 Albuquerque, New Mexico

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Denver, Colorado

4.2.2 Groundwater Pumping

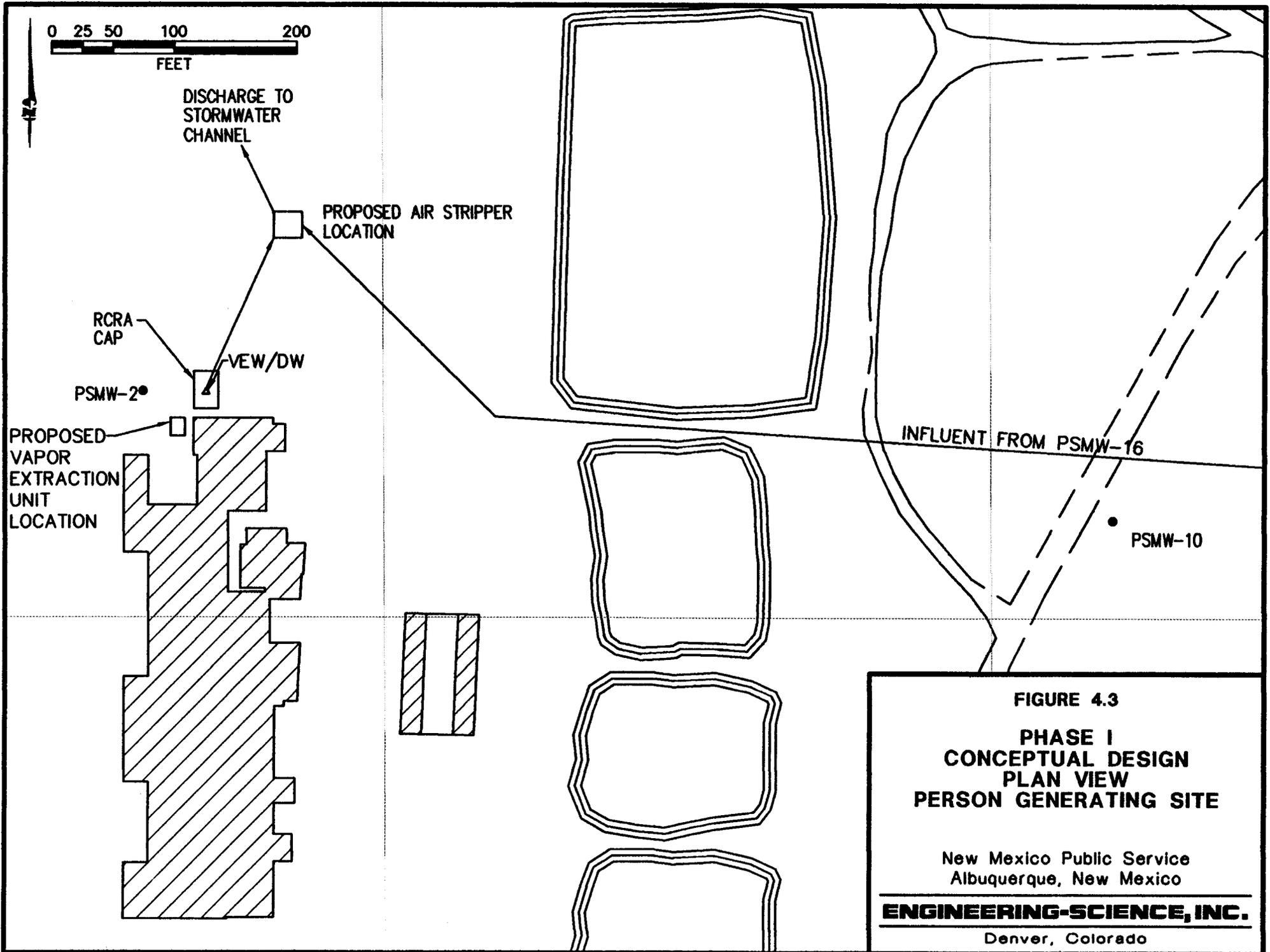
Phase I groundwater pumping tests are required to determine the rate at which contaminated groundwater can be extracted from the aquifer and to estimate the capture zone for each well in a full-scale design. An additional benefit of these tests will be to optimize the removal efficiency of groundwater treatment systems. Long-term pumping tests will be conducted at two locations. Existing monitoring well PSMW-16 will be used to demonstrate pumping efficiencies in highly permeable aquifer material. The dual-purpose well (VEW/DW) near the former waste tank (shown in Figure 4.1) will be used for both soil vapor extraction and Phase I pumping tests. Both of these wells will be/are constructed of 4-inch PVC with screened intervals extending approximately 15-20 feet below the groundwater. Initial pumping tests conducted by METRIC Corporation indicated that these two wells would be located in two distinct zones of low (VEW/DW) and high (PSMW-16) hydraulic conductivity and will provide a good estimate of long-term pumping performance. These wells are also located in the most contaminated areas of the solvent plume and will provide a conservative test of groundwater treatment capabilities.

Pumping tests will be conducted for approximately 60 days at each well using submersible pumps lowered approximately 15-20 feet below the initial groundwater level in each well. The sustained pumping rate and drawdown at each well will be recorded and the capture zone estimated by measuring drawdown in surrounding monitoring wells. Groundwater samples will be taken regularly from each well discharge throughout the 60-day pumping test to better estimate the rate of contaminant removal. After the pumps have been off for approximately 30 days, the pumps will be restarted and extracted groundwater resampled to determine the rate at which adsorbed contaminants reequilibrate with the groundwater. This data will be used to determine the potential efficiencies of pulsed pumping operations during Phase II.

4.2.3 Groundwater Treatment

Air stripping has been selected as the most efficient and cost effective method of removing volatile chlorinated solvents from groundwater. A skid-mounted, low-profile air stripping unit will be used during the 60-day pumping test to treat an estimated 40 gpm from the two test wells. A plan view showing the potential location of the air stripper and other systems is shown in Figure 4.3. Based on April 1993 monitoring data from these wells and short-term pumping tests, flow rates of approximately 2 gpm from the new VEW/DW and 35-40 gpm from PSMW-16 are predicted. The flow-averaged initial contaminant concentrations expected to enter the air stripper are shown in Table 4.1.

Based on this contaminant loading, PCE will be the most difficult to remove due to its higher influent concentration. The air stripper must be capable of reducing the average PCE influent concentration of approximately 100 ppb to the target MCL of less than 5 ppb. A low-profile, tray air stripping unit is recommended for this application. These systems are mechanically simple, reliable, and easier to maintain than packed-tower air strippers. Based on the flow-averaged influent estimate presented in Table 4.1, air stripper manufacturers have recommended several treatment system designs which will achieve effluent concentrations of less than 5 ppb for all contaminants. Information on



4-6

FIGURE 4.3
PHASE I
CONCEPTUAL DESIGN
PLAN VIEW
PERSON GENERATING SITE

New Mexico Public Service
 Albuquerque, New Mexico

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 Denver, Colorado

one of the candidate air stripping units is provided in Appendix A. This unit is capable of treating up to 50 gpm to the required MCLs for all contaminants. During Phase I, the efficiency and reliability of the air stripping unit will be evaluated. A second unit could be installed to treat the additional groundwater removed during Phase II plume remediation.

TABLE 4.1
INITIAL CONTAMINANT CONCENTRATION IN TEST WELLS

Well	Concentration ($\mu\text{g/L}$) ^{a/}				Estimated Flow (gpm)
	PCE	DCE	TCA	Total	
VEW/DW	150	70	36	256	1
PSMW-16	100	58	6	164	34.8
Estimated Flow Averaged Influent	101	58	7	166	35.9

^{a/} Based on April 1993 sampling event.

4.2.4 Discharge of Treated Water

Phase II treatment is expected to produce 50 to 70 gpm of water which meets drinking water standards. Phase I testing will produce up to 40 gpm. The treatment system is expected to produce an effluent of <5 ppb PCE, <5 ppb of 1,1-DCE, and <5 ppb of 1,1,1-TCA.

Two options for effluent discharge are under consideration. The first option would be to pump treated groundwater for use in irrigation systems. This beneficial use of the water would reduce the quantity of groundwater pumped for local irrigation and provide a continuous source of high quality water for many years. While this option has the obvious benefit of conserving water resources, thousands of feet of underground pipeline and a transfer pump may be required to convey the water to a point of beneficial use.

The second option is to directly discharge water to the stormwater drainage system northwest of the site. This discharge to the AMAFCA South Diversion Channel would eventually reach the Rio Grande River. Although there is little construction associated with this option, it will require an update and rapid approval of the current NPDES permit application.

A hybrid of these options would be to discharge to stormwater drainage for the 60 days of Phase I testing with the beneficial use option developed for long-term Phase II discharge.

4.2.5 System Operation and Monitoring

A preliminary schedule for Phase I operations is provided in Figure 4.4. Phase I operations will consist of approximately 30 days of equipment installation, startup, and optimization and an additional 60-70 days of intensive soil vapor extraction in the source area and extended pumping of the new VEW/DW and PSMW-16. During the first 30 days, the VEW/DW will be constructed, submersible pumps will be installed in VEW/DW and PSMW-16, and the piping manifold to the air stripping system will be constructed. This construction will be followed by the startup of the vapor extraction unit and groundwater pumping and treatment systems. Monitoring of these systems will be most intense during the initial weeks of operation and less frequent as the system influent and effluents stabilize. A schedule of the recommended monitoring for Phase I is provided in Table 4.2.

4.3 PHASE II CONCEPTUAL DESIGN

4.3.1 Use of Phase I Results

The design for Phase II pumping operations will not be finalized until the completion of Phase I testing. Information from Phase I, such as sustained pumping rates, the capture zone of each well, contaminant concentrations, air stripper performance and the benefits of pulsed pumping, will be examined during Phase I. This data will aid in the optimization of the design and operation of the groundwater treatment system and reduce total project costs. The following sections describe our preliminary recommendations for continued soil vapor extraction in the source area and long-term plume remediation.

4.3.2 Continued Soil Vapor Extraction

Due to the low cost of soil vapor extraction and its ability to remove significant contaminant mass from the subsurface, ES recommends that this technology continue to remediate source area soils until contaminant recovery is reduced to levels of approximately 1 ppmv total volatiles. The total vapor extraction rate from the Phase II system is estimated at 100-150 scfm. A more permanent 15-HP blower unit will replace the pilot blower for extended use. The concentration of volatile organics in this air stream should be low enough to eliminate the need for vapor-phased carbon treatment during Phase II.

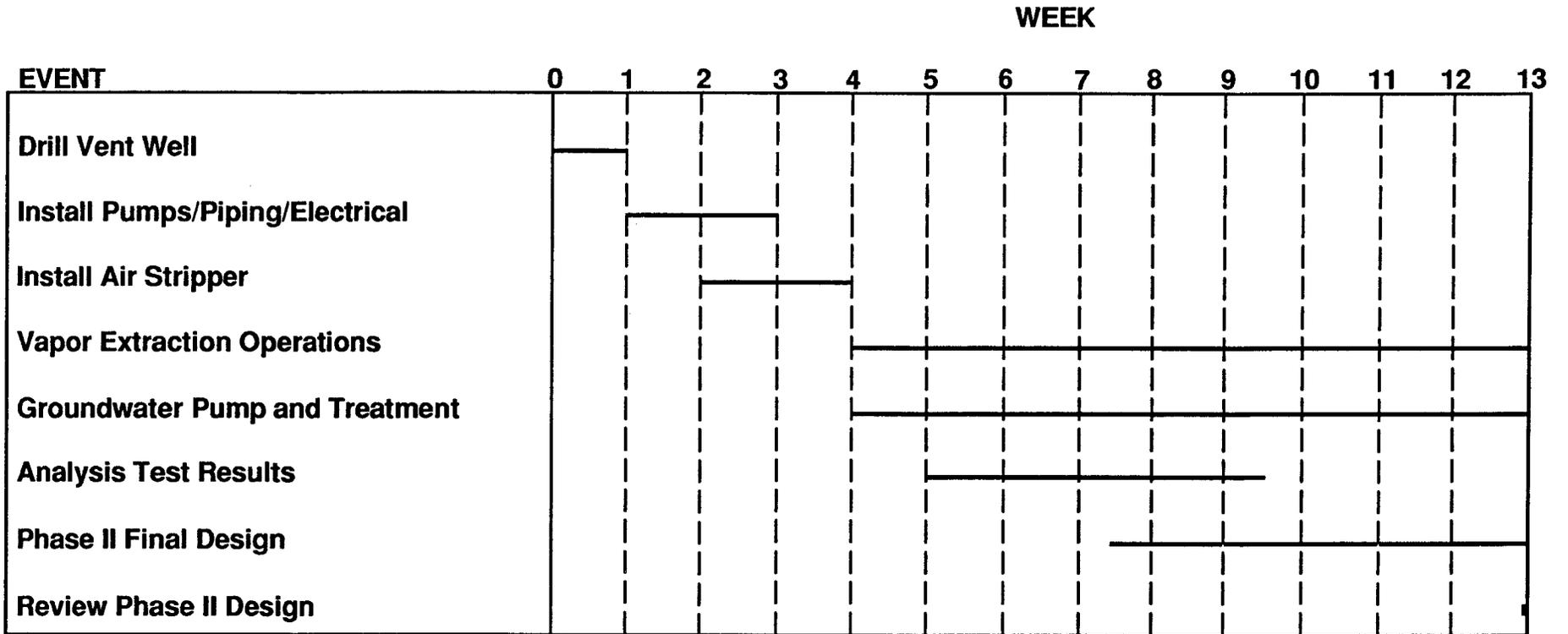


FIGURE 4.4

**PROPOSED PHASE I SCHEDULE
SOIL AND GROUNDWATER
REMEDICATION AT
PERSON GENERATING SITE**

New Mexico Public Service
Albuquerque, New Mexico

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Denver, Colorado

TABLE 4.2
PROPOSED MONITORING SCHEDULE
FOR PHASE I OPERATIONS

Media/Location	Type of Analysis	Method	Frequency
Groundwater Influent from VEW/DW/PSMW-16	Volatile Organics	EPA601/8010	one/week/well first month, monthly thereafter
Treated Groundwater	Volatile Organics	EPA601/8010	two/week first month, biweekly thereafter
Groundwater wells VEW/DW, PSMW-16	Water Levels	Electronic Water Level Indicator	hourly first day, twice daily first week, weekly for first month, after 60 days
Influent Soil Gas from VEW	Specific Volatile Organics	EPA TO-14	daily first week, weekly thereafter
	Total Organics Volatile	Handheld PID*	
Effluent from activated carbon treatment vapor unit	Specific Volatile Organics	EPA TO-14	daily first week, weekly thereafter
	Total Volatile Organics	Handheld PID	

*Photo Ionization Detector

4.3.3 Estimated Plume Pumping Network

Although the exact number and location of Phase II plume remediation wells will be determined after Phase I testing, an estimated 4 to 6 extraction wells will be required to remove groundwater currently exceeding 5 ppb of PCE and 1,1-DCE. Section 5 describes the results of preliminary groundwater modeling and the projected fate and transport and risk of plume contaminants. Initial groundwater modeling of this plume predicted that a series of four extraction wells located along the plume centerline west of Interstate 25 would be capable of reducing concentrations of all VOCs below the 5 ppb MCL after six years of continuous pumping and 14 years of natural attenuation. Following Phase I testing, updated pumping information will be used in the model simulation to finalize the Phase II extraction well locations.

4.3.4 Groundwater Treatment

The performance of the Phase I air stripping system will be used to finalize the design and procurement of the expanded Phase II groundwater treatment system. An estimated 50-70 gpm will be extracted during Phase II operations. The concentrations of contaminants in this groundwater should be significantly lower than concentrations

encountered during Phase I for two reasons. First, Phase I pumping will take place in the most contaminated areas of the plume while Phase II will include downgradient areas with much lower initial concentrations. Second, the concentration of recovered contamination at all wells will decrease over time.

The conceptual design for Phase II groundwater treatment has two options. The first option would continue to use the Phase I 50 gpm air stripping system and purchase a second identical system to handle the increased Phase II flow rate. The second option would be to rent the Phase I air stripper for the 60-90 day test and then replace it with a larger, single air stripping system for long-term operations. The most economical and best operation and maintenance alternative will be selected.

Monitoring of the Phase II groundwater treatment system will follow a similar schedule as Phase I with the most intense influent and effluent sampling during the initial weeks of operation and less frequent sampling as the system influent and performance stabilizes. The frequency of long-term sampling will be determined by the discharge permit, or requirements of the user if beneficial irrigation can be arranged.

4.3.5 Discharge of Treated Water

If technically and economically feasible, the treated groundwater from long-term pumping operations could be used for some beneficial purpose such as landscape or recreational irrigation. PNM and its contractors will work to establish a beneficial use and evaluate the costs of these options. Because irrigation requirements are seasonal and may not require a continuous supply at 50-70 gpm, attempts to finalize a NPDES discharge permit should be made to insure that an approved discharge option always exists.

4.3.6 System Operation, Maintenance, and Monitoring

The soil vapor extraction and air stripping systems specified for long-term operation will be mechanically simple and engineered for reliable, low-maintenance operation. It is anticipated that the soil vapor extraction system will operate only 18-24 months. ES has operated similar systems for this period of time with minimal downtime. Routine maintenance includes a monthly change of the unit's air filter and removing any condensate that may have accumulated in the knock-out pot (see Figure 4.2). During the initial months of operation, maintenance may also include replacement of carbon canisters and shipment of spent canisters to the manufacturer for regeneration.

The trays on the proposed air stripping system have been designed for easy cleaning. The cleaning schedule for the air stripper will be a function of the groundwater chemistry. Iron precipitation and mineral scaling can slowly foul and reduce the performance of air stripping systems. Monthly cleaning is anticipated and can be accomplished during the same day that the soil vapor extraction system is being checked.

Phase II monitoring will be designed to comply with the Post-Corrective Measures Implementation Report and to determine the performance of three primary systems: contaminant removal from the aquifer, contaminant removal in the air stripping system,

and contaminant removal in the soil vapor extraction system. The monitoring frequency and type of analysis recommended during Phase II is presented in Table 4.3.

TABLE 4.3
RECOMMENDED MONITORING SCHEDULE FOR
PHASE II OPERATIONS

Location	Type of Analysis	Method	Frequency
CAD Designated Groundwater Monitoring Wells	Volatile Organics	EPA601/8010	Semi-Annually*
Air Stripper Influent	Volatile Organics	EPA601/8010	Monthly*
Air Stripper Effluent	Volatile Organics	EPA601/8010	Monthly*
Vapor Extraction Influent	Volatile Organics	EPA TO-14	Monthly
During Activated Carbon Treatment Effluent	Volatile Organics	EPA TO-14	Weekly

* Sampling will proceed from monthly, to quarterly, to semi-annually according to the schedule of the Post Corrective Measures Implementation Report.

SECTION 5

ANALYSIS OF EXPECTED PERFORMANCE AND RISKS

In order to investigate the role of natural fate and transport processes in minimizing the risk of contamination from the Person Generating Station site, ES modeled the behavior of the VOC groundwater plume under several different remedial alternatives. The intent of this study was to use a fate and transport model based on available site data to estimate (1) the effectiveness of natural physical and chemical processes in reducing contaminant concentrations and (2) the duration of the pump-and-treat action at the Person Generating Station site to achieve and maintain shallow groundwater concentrations less than 5 ppb for an indicator, persistent contaminant of concern (PCE). ES anticipates that natural fate and transport processes may be quite effective in minimizing both the required project implementation time and the potential risk of exposure from any contaminants of concern at the site. The purpose of this initial fate and transport modeling was to help define the relative contributions of certain natural physical and chemical processes and aquifer pumping on the total remediation process. A total of three remediation scenarios were modeled, including the source removal/no extended pumping scenario. The results of this modeling have provided valuable information on the placement of groundwater pumping wells, the pumping time required to approach MCLs, and the present and future risk of remaining contaminants.

5.1 FATE AND TRANSPORT MODEL

The behavior and fate of the contaminants of concern in the shallow groundwater underlying the Person Generating Station site is controlled by both physical and chemical processes characteristic of both the compounds and the aquifer. Physical transport processes such as advection, dispersion, and molecular diffusion play a significant role in determining the behavior and fate of VOCs in the aquifer. However, if these were the only processes influencing the movement of the VOCs, the shallow groundwater plume would be much larger than monitoring data suggest. The physical and chemical properties of the contaminants also govern transport behavior. For example, chemical processes such as hydrolysis (the reaction of VOC molecules with water molecules to form other compounds), degradation (the transformation of VOCs into other compounds, which can sometimes be mediated by microorganisms), and sorption (absorption into and adsorption onto) to immobile aquifer media can have a significant impact on the long-term concentration of contaminants in the groundwater. These natural attenuation processes can significantly reduce contaminant migration and retard the spread of the VOC groundwater plume.

ES has investigated the potential impact of natural processes on the spread of the VOC groundwater plume in order to assess any potential risk of exposure from the contaminants under several groundwater pumping scenarios. A finite-difference groundwater model (MODFLOW) was coupled to a solute transport model (MT3D) to simulate contaminant behavior in an aquifer under three different scenarios: source removal/no long-term pumping alternative, the influence of source removal and five steady-state pumping wells for 9 years (alternative 2), and the influence of source removal and four steady-state pumping wells for 6 years (alternative 3). The model was developed using site-specific assumptions about governing natural physical and chemical processes. The reduction in contaminants due to natural attenuation is expected to exceed model predictions based on the natural reductions which have occurred over the past decade.

5.1.1 Conceptual Model Development

MODFLOW, a three-dimensional groundwater flow model developed and distributed by the U.S. Geological Service, was used to simulate groundwater flow at the Person Generating Station site. This model is capable of simulating hydrologic phenomena that influence groundwater movement, such as pumping wells in a heterogeneous aquifer. Because it is a three-dimensional model, MODFLOW is also capable of simulating groundwater flow in a multi-layered, multi-aquifer system.

The MT3D model is a three-dimensional groundwater contaminant transport model to be used in conjunction with a groundwater flow model such as MODFLOW. The groundwater flow model is calibrated independently of the transport model. MT3D retrieves the hydraulic heads developed by the flow model and automatically incorporates the flow field into the transport model. MT3D allows for the spatial variance of dispersion, sorption, and adding other sources and sinks (e.g., groundwater extraction wells). Additional information on these models is provided in Appendix B.

Data collected at the site indicate that contamination is confined to the upper 20 feet of the water table aquifer. For this reason a three-layer model in the vertical direction was developed that separates the upper contaminated zone from the underlying zone. The second layer in this three-layer model is a thin layer used as a buffer between the upper and lower zones. The third and final layer is a thick flow zone representing the deeper portion of the aquifer. Vertical hydraulic conductivities were set at approximately 10 percent of the horizontal hydraulic conductivities for all three layers.

Constant head boundaries for layers one, two, and three were set on the west and east end of the modeled area. The elevation of the constant heads was set far away from the site so that site activities would not affect the boundary conditions. No flow boundaries were placed on the north and south end of the modeled area. The other boundary consisted of the placement and operation of the various extraction wells used to implement the remediation scenarios. These wells were assumed to be screened in the uppermost 20 feet of the aquifer.

PCE was selected as the model's indicator contaminant for several reasons. First, the existing PCE plume is larger in areal extent than the other contaminants. Second, PCE is the contaminant most adsorbed to saturated soils. As a result, PCE may be more difficult to recover using pump-and-treat activities. Modeling PCE provides a

worst-case scenario for total time required to remove VOC contamination from the aquifer using a pump-and-treat approach. Third, PCE cannot be readily hydrolyzed or biodegraded in this aerobic aquifer. Thus, PCE is the most persistent VOC present in the shallow aquifer. The model code did not include any potential natural removal reactions such as hydrolysis or degradation. Finally, the model code did not account for the removal of PCE from partitioning onto soils which become unsaturated as the water table continues to fall. Thus, a relatively conservative model approach was developed using a persistent contaminant to predict "worst-case" treatment requirements under the various remedial scenarios.

5.1.2 Input Parameters

The Person Generating Station site was divided into a three-dimensional grid system consisting of columns, rows, and layers. Horizontal hydrogeologic properties of the site were defined by 51 rows and 71 columns varying in width from 100' to 800'. The closer spacing was used on the site and the wider spacing was used further from the site.

Aquifer dimensions and hydraulic parameters used in the model were obtained from field measurements presented in the CAD assessment (METRIC, 1993). The spatial distribution of the hydraulic conductivity was based on the results of short-term pump tests at 26 of the site groundwater monitoring wells. In areas where variables were uncertain, the input variables were extrapolated in a general north-south alignment from the site. This north-south orientation corresponds to the general structural trend of the Rio Grande Basin. Summary input values for the model are presented in Table 5.1; further details on model input parameters can be found in Appendix B.

The present distribution of PCE concentrations as shown in Figure 1.2 was used as initial conditions for the contaminant transport model (METRIC, 1993). The longitudinal dispersivity used for modeling is 30 feet, with transverse and vertical dispersivities of 3 feet each. These values should underestimate expected dispersion effects, thus predicting the highest concentration levels to be expected under steady-state flow conditions. Sorption of the indicator contaminant PCE was incorporated into the model code using a retardation factor. A retardation factor is a measurement of how slowly a contaminant moves through the aquifer in comparison to mass groundwater movement. For this study, a uniform retardation factor equal to 1.03 was calculated using a uniform bulk density appropriate for PCE in sandy soils (Freeze and Cherry, 1979) and a literature distribution coefficient (Montgomery and Welkom 1990); these calculations are presented in Table 5.2. Use of this retardation factor will overestimate the extent of VOC plume migration over the next 20 years under each remedial scenario but could somewhat underestimate the time necessary to implement full treatment using pump-and-treat technologies. A sensitivity analysis using a retardation factor of 2.0 was also completed to determine its input on pumping time.

5.1.3 Model Calibration

Using available hydraulic conductivity data for the site, the groundwater flow model was calibrated until the potentiometric (groundwater table elevation) measurements in the simulated sand aquifer correlated reasonably well to the potentiometric data at the Person Generating Station site (METRIC, 1993). The MT3D model was also

TABLE 5.1
MODEL INPUT PARAMETERS

	Layer 1	Layer 2	Layer 3
Longitudinal Hydraulic Conductivity (ft/day)	1-200	1-200	1-200
Transverse Hydraulic Conductivity (ft/day)	1-200	1-200	1-200
Vertical Hydraulic Conductivity (ft/day)	1-20	1-20	1-20
Storativity	0.15	0.15	0.15
Porosity	0.3	0.3	0.3
C _o	Existing VOC plume concentrations	-	-
Longitudinal Dispersivity (feet)	30	30	30
Vertical Dispersivity (feet)	3	3	3
Transverse Dispersivity (feet)	3	3	3
PCE Retardation Factor	1.03	1.03	1.03

TABLE 5.2
CALCULATION OF RETARDATION FACTOR FOR PCE

Equation:
$$R = 1 + \frac{\rho_b K_d}{n_e} = 1.03 \text{ for PCE}$$

where:

- R = retardation factor
- ρ_b = bulk density (1.65 g/cm³)
- K_d = partition coefficient (cm³/g)
- n_e = effective porosity (0.3)

and:

$$K_d = K_{oc} f_{oc}$$

where:

- K_{oc} = organic carbon partition coefficient (1.81 cm³/g)
- f_{oc} = fraction of organic carbon (0.0027)

calibrated and verified to historical contamination data by developing transport parameters that can be used to simulate existing aquifer conditions. Once the model was calibrated, 20-year simulations were run under three possible remediation scenarios. Details on model calibration appear in Appendix B. Results from subsequent modeling runs were plotted and contoured on a Person Generating Station site map; these results and accompanying discussions appear below.

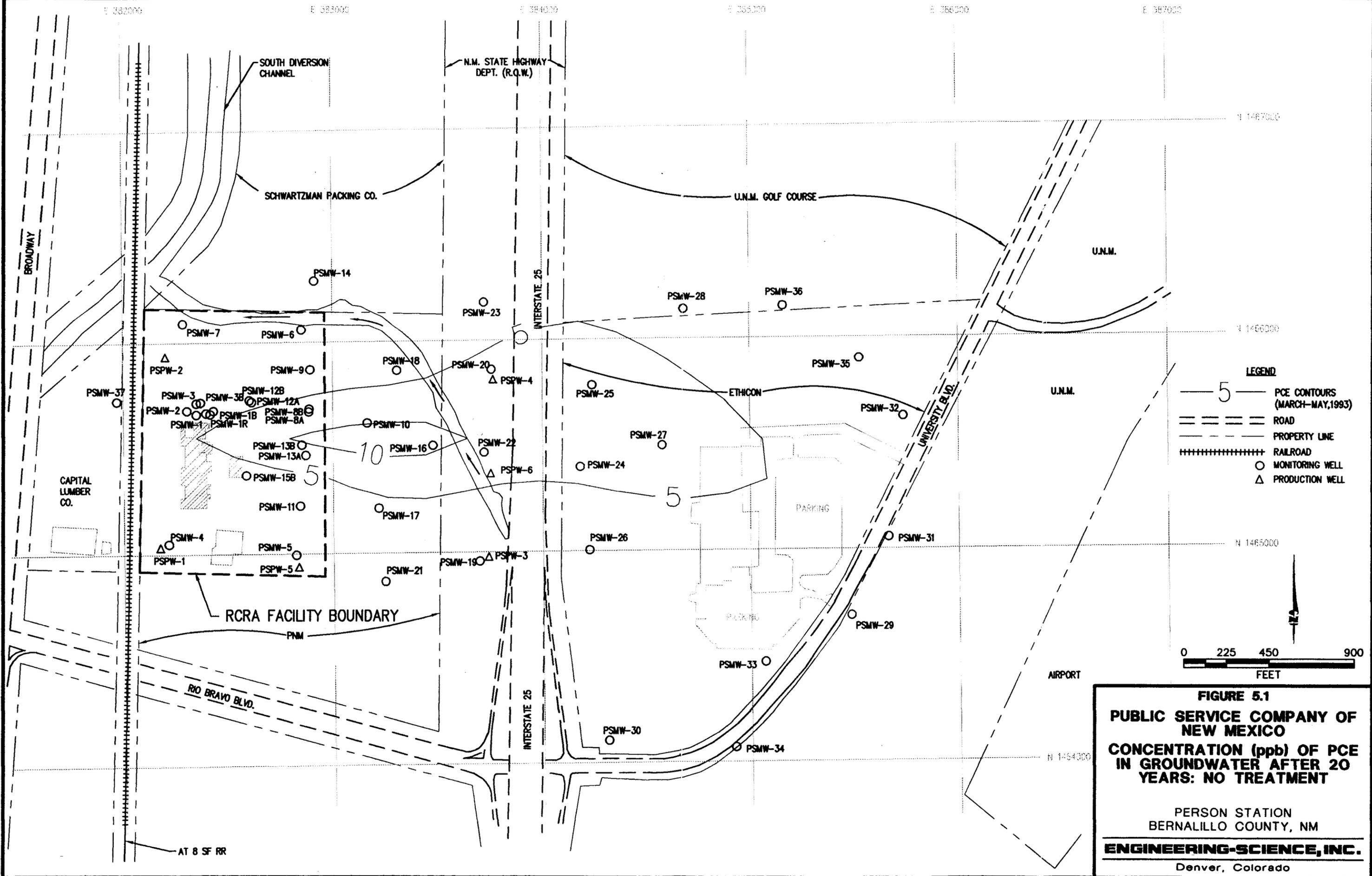
5.2 QUANTITATIVE EVALUATION OF REMEDIAL ALTERNATIVES

This initial modeling effort was designed to investigate the possible behavior of the shallow groundwater VOC plume under the Person Generating Station site if: (1) only source removal was completed without plume pumping, 2) source removal and a pump-and-treat program using five groundwater extraction wells on the Person Generating Station property was employed 9 years, and (3) source removal and a reduced pump-and-treat program using only four groundwater extraction wells was implemented for 6 years. An estimate of the size and location of the VOC plume after 20 years for each of these different scenarios has provided valuable information as to the required nature and scope of the remediation activities for the site. Further, estimated contaminant residual concentrations can be used to estimate the potential risk to humans and other ecological receptors in the area and be used to support risk-based decisions on remediation activities appropriate for the Person Generating Station site.

5.2.1 Alternative 1: Source Removal Only

The purpose of modeling this scenario was to provide comparative baseline information on the foreseeable extent and rate of contamination migration without any pumping downgradient of the source area. The behavior of the indicator contaminant of concern was modeled for a period of 20 years in the absence of any groundwater pump-and-treat action outside of the source area.

Using the steady-state flow field derived from the calibrated MODFLOW/MT3D model, the fate and transport of the existing PCE shallow groundwater plume (as shown in Figure 1.2) was investigated under the no plume pumping scenario, which assumes no additional sources that contribute to groundwater contamination are present. This simulation assumes that the source area soils and shallow groundwater are remediated through soil vapor extraction and shallow groundwater pumping to prevent additional VOCs from entering the plume. The results of the modeling study are presented in Figure 5.1. These data suggest that after 20 years, the concentration of PCE will have decreased to approximately 10 ppb in the most contaminated portions of the shallow groundwater plume. Further from these areas, the concentration of PCE almost achieves the target MCLs due to the effects of dispersion. These data also indicate that the areal extent of the VOC groundwater plume will widen slightly but remain in approximately the same position as the current plume. These modeled predictions seem to be supported by the groundwater analytical data collected in the CAD assessment which indicate a general stabilization in the VOC plume.



5.2.2 Alternative 2: Source Removal and Implementation of a Groundwater Pump-and-Treat System for 9 Years

This alternative assumed source removal and implementation of a pump-and-treat system in the area containing the highest concentration of groundwater contamination. The MODFLOW/MT3D simulation included five extraction wells located along the centerline of the plume source area on the Person Generating Station property. The combined extraction flow rate for the five wells is approximately 60-70 gallons per minute. The model assumed that groundwater extraction would continue for a period of 9 years with the intent of reducing PCE to concentrations approaching the target MCLs throughout the entire plume. At the end of this time period, the pumps were shut down, allowing the remaining contaminants to attenuate through natural processes for an additional 11 years. Figure 5.2 illustrates the distribution of PCE after the 20 year period using this pumping scenario. This model scenario predicts a reduction in the concentration of PCE to below target MCLs.

5.2.3 Alternative 3: Source Removal and Implementation of a Groundwater Pump-and-Treat System for 6 Years

This modeling effort was designed to compare the effectiveness of a 6-year groundwater pumping project using only four extraction wells with that demonstrated for the 9-year project using five extraction wells. Four extraction wells located along the plume centerline on the Person Generating Station property would be pumped at a combined rate of 60 gallons per minute for a total of 6 years. In this scenario, pumping would begin immediately and cease after 6 years, leaving a remaining (maximum) plume concentration of about 7 ppb of PCE. Natural fate and transport processes in the subsequent 14 years would then be able to reduce PCE concentrations to below target MCLs as shown in Figure 5.3. Figure 5.4 illustrates the predicted decrease in monitoring well concentrations over time using this pumping scenario.

5.3 MODEL CONCLUSIONS AND SENSITIVITY ANALYSIS

Based on these model simulations, it is possible that implementation of a pump-and-treatment action using four extraction wells located along the plume centerline for 6 years should be sufficient to treat shallow groundwater VOC contamination to levels at or below target MCLs for all three of the contaminants of concern within 20 years. Implementation of a five-well system for 9 years would also reduce VOC concentrations to target MCLs.

Some of the issues inherent in developing an adequate model for any site is the development of input parameters that simulate site-specific conditions. Two of the most important parameters in this model simulation are vertical hydraulic conductivity values and retardation values. High vertical hydraulic conductivity values will increase vertical dispersion, thus decreasing the rate and extent of horizontal plume migration. In contrast, low retardation factors can overestimate the rate and extent of horizontal plume migration and the effectiveness of any pump-and-treat activity on the shallow groundwater plume. The possible influences of these parameters on model simulations can be evaluated separately. The effect of changing the vertical hydraulic conductivity was evaluated quantitatively using the calibrated MODFLOW/MT3D model pumping

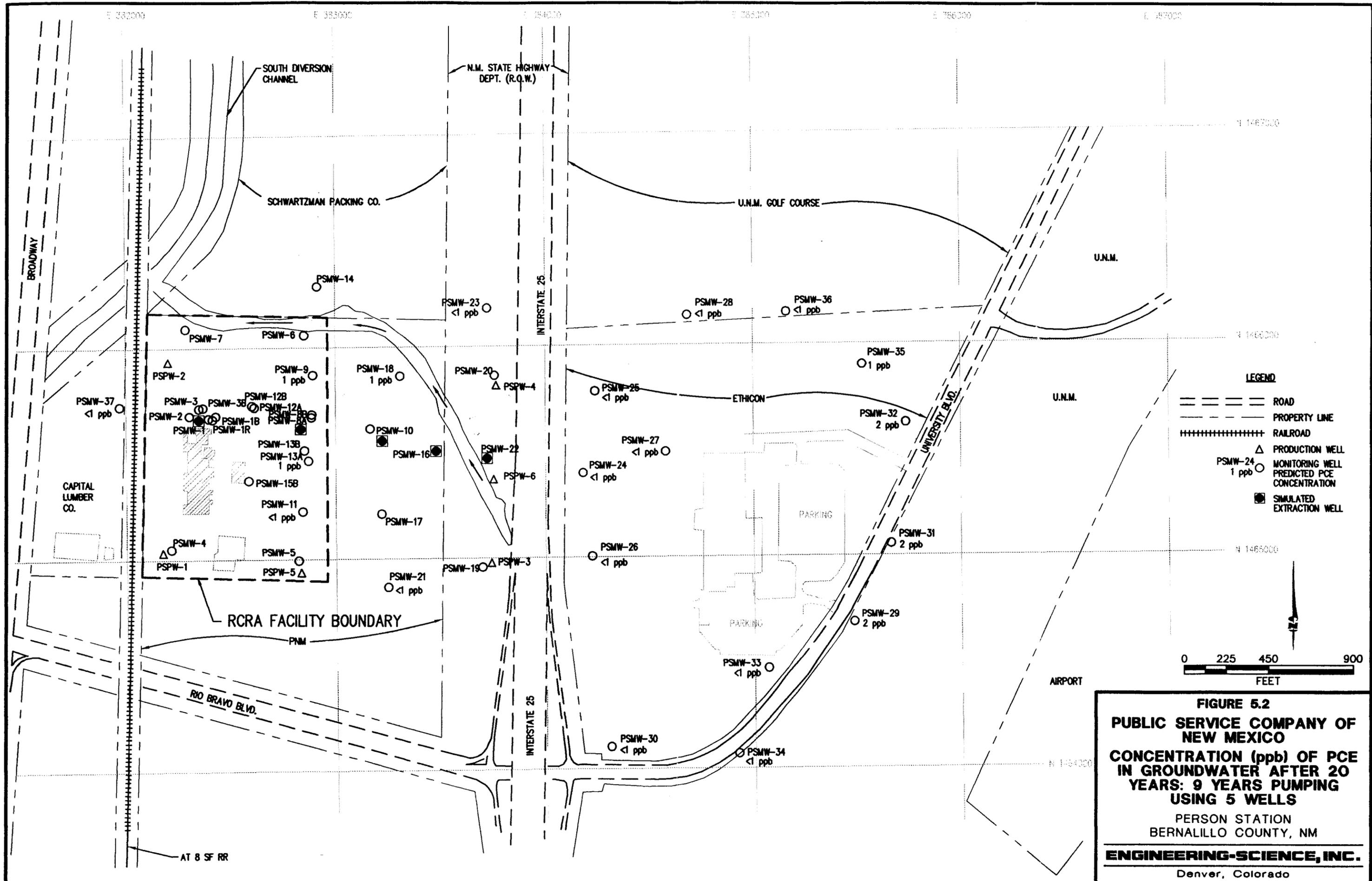
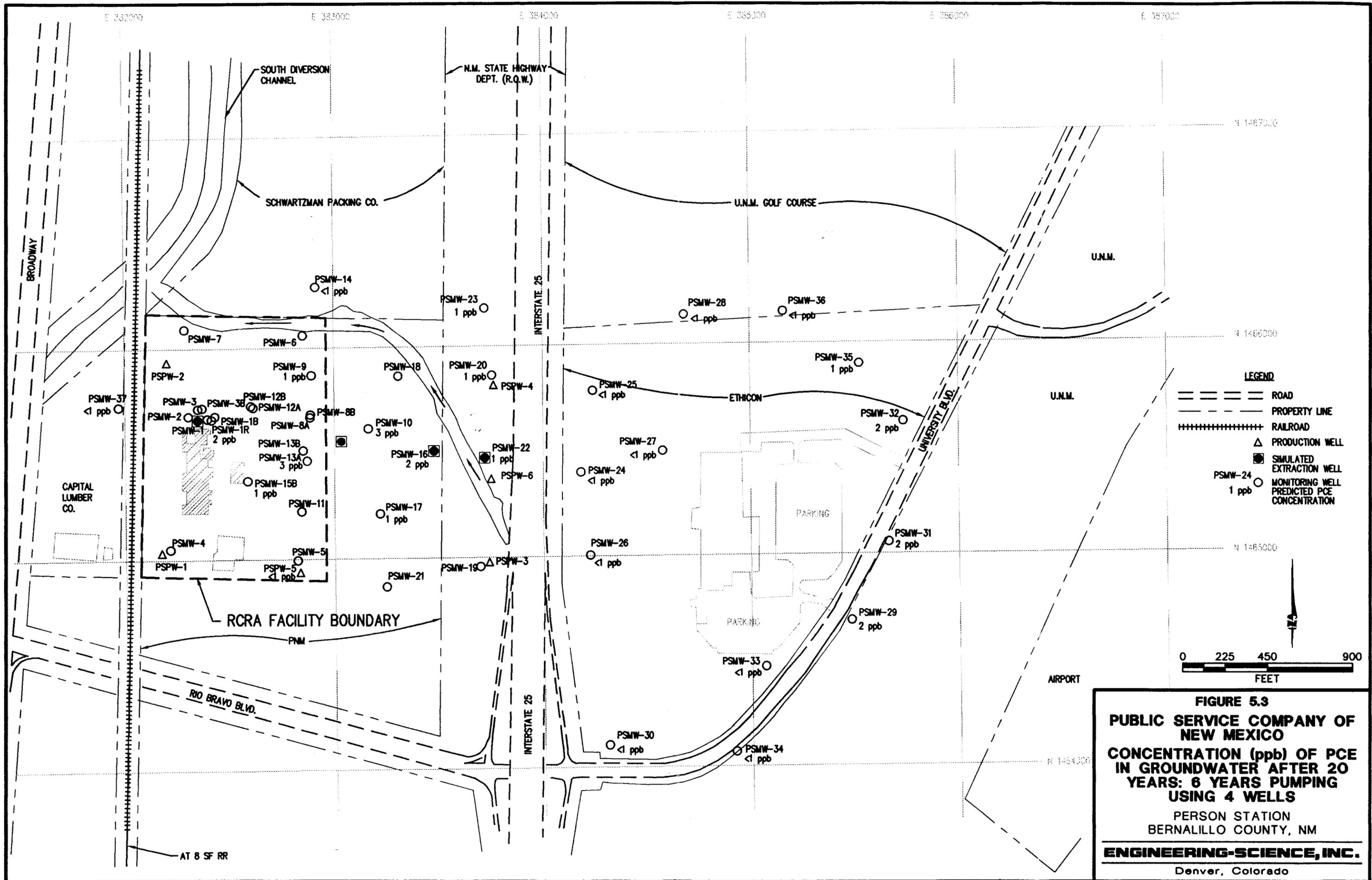


FIGURE 5.2
PUBLIC SERVICE COMPANY OF
NEW MEXICO
CONCENTRATION (ppb) OF PCE
IN GROUNDWATER AFTER 20
YEARS: 9 YEARS PUMPING
USING 5 WELLS
 PERSON STATION
 BERNALILLO COUNTY, NM
ENGINEERING-SCIENCE, INC.
 Denver, Colorado



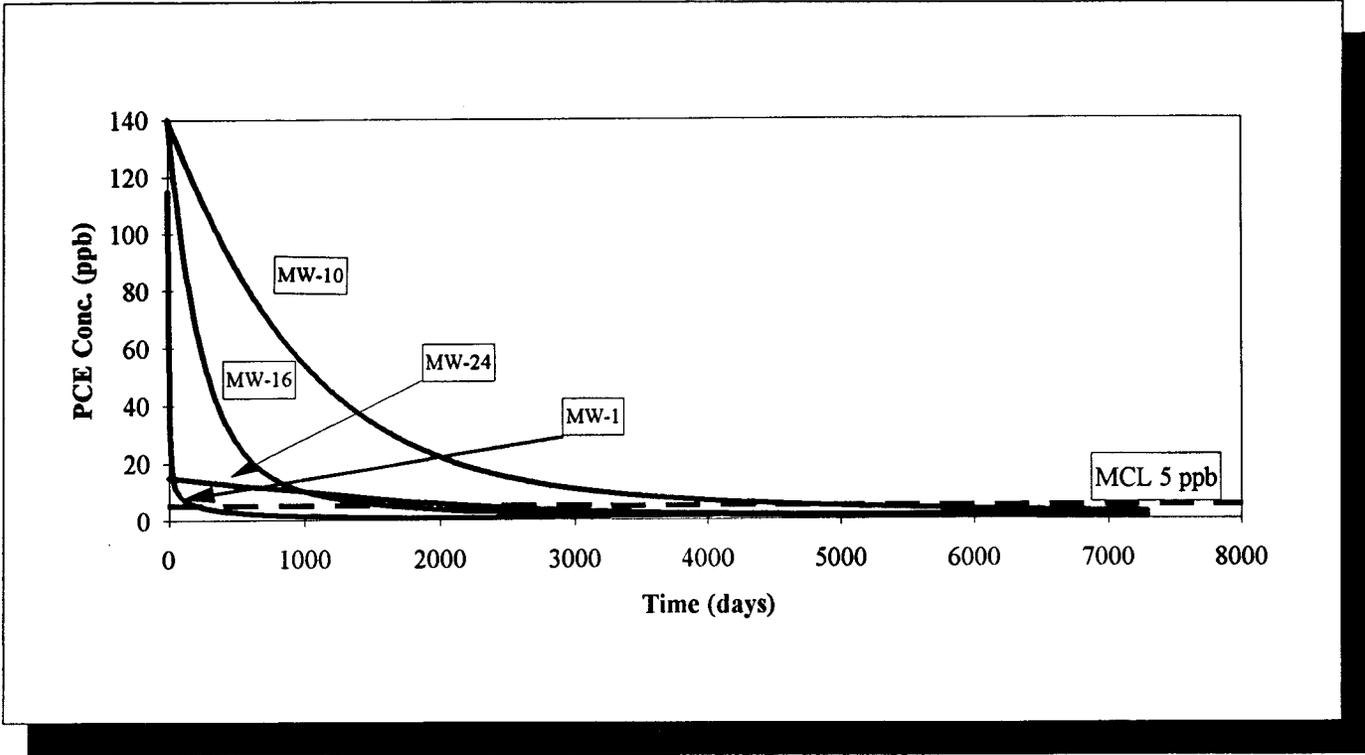


FIGURE 5.4
PUBLIC SERVICE COMPANY OF
NEW MEXICO
PREDICTED CONCENTRATIONS (ppb)
OF PCE IN GROUNDWATER AFTER
20 YEARS: 6 YEARS PUMPING
USING 4 WELLS
PERSON STATION
BERNALILLO COUNTY, NM

ENGINEERING-SCIENCE, INC.

Denver, Colorado

for 6 years using four extraction wells. The vertical hydraulic conductivity was decreased from 10 percent to 1 percent of the horizontal hydraulic conductivity ft/day and the model was used again to simulate 6 years of pumping using four extraction wells. Results from this order of magnitude change in the vertical hydraulic conductivity are not significantly different from those illustrated in Figure 5.3. Influences from changing the retardation factor were also considered. If the retardation factor is increased from 1.03 to 2.0, which would increase the adsorption of PCE (e.g., higher retardation factor), the overall effectiveness of the 6 year pumping program should be decreased. The sensitivity analysis using a retardation factor of 2.0 is included in Appendix B. This scenario shows a small area of the plume may slightly exceed the 5 ppb MCL, and that additional pumping time may be required immediately downgradient of the source area.

The purpose of this preliminary modeling effort is to begin to estimate the relative contributions of natural attenuation and pumping on contaminant removal and to evaluate the potential for significant risk due to contaminated groundwater migration. The model will be re-calibrated as necessary following Phase I pumping and will be used to design the Phase II pumping system. Further details on model calibration, input parameters, and uncertainties can be found in Appendix B.

5.4 EXPOSURE ASSESSMENT

A contaminant entering the environment will cause adverse effects if (1) it exists in a form and concentration sufficient to cause harm, and (2) it comes in contact with organisms or environmental media with which it can interact. In general, the goal of an exposure assessment is to determine the type and magnitude of potential exposures to contaminants of concern that are present at or migrating from a site. This information can then be coupled with chemical-specific toxicity information to characterize potential site risks.

As part of the exposure assessment, potential receptors to site-related contamination and the pathways through which they might be exposed are identified. Pathways are evaluated for the probability of completion. If a pathway is not completed, there is no risk. As the following discussions will show, none of the pathways of exposure identified at the Person Generating Station site are currently completed. Evaluation of potential future pathways of exposure may be conducted upon completion of site remedial activities.

It is not the intent of this report to conduct a quantitative risk assessment in compliance with all USEPA guidance materials. Rather, the report focuses on identifying potential exposure risks that may exist at the site using only appropriate model-derived concentrations and chemical toxicity data. If a potential risk were identified at this time, it would be appropriate to consider additional, detailed investigations.

An evaluation of the environmental fate and transport properties of the contaminants of concern for the Person Generating Station site can help determine the potential for migration in the environment and the potential for receptor exposure to the contaminants. The environmental fate and transport of contaminants is dependent upon

the physical and chemical properties of the compounds, the environmental transformation processes affecting them, and the media through which they migrate. In the following sections, the relevant chemical and physical properties of the contaminants of concern are presented and discussed in relation to specific environmental media pathways. Preliminary conclusions about the potential for completion of any one specific pathway are presented.

5.4.1 Soils Pathway

The source waste oil tank was removed and closed in 1987 in accordance with an approved RCRA Closure Plan. Site assessment data indicate that approximately 60,000 cubic feet of near-surface soils currently containing at least 1 ppm PCE remain at the Person Generating Station site. Although PCE, 1,1-DCE, and 1,1,1-TCA are present at relatively low concentrations in near-surface soil media, higher concentrations of these compounds have been shown to result in adverse noncarcinogenic health effects. For example, both oral and inhalation exposure to high concentrations of 1,1,1-TCA can cause adverse effects; however, the only exposure pathway causing adverse health effects for PCE and 1,1-DCE is oral ingestion of contaminated material. Note that the USEPA considers 1,1-DCE to be a carcinogen through both oral and inhalation routes. However, pathways of exposure to these contaminants in soils at the Person Generating Station site are not currently complete through either oral or inhalation routes of exposure. Dermal and oral exposure to the contaminated near-surface soil is unlikely due to the permanent concrete cap placed over the source area. Therefore, there is no current risk from this pathway. However, all three contaminants of concern also are relatively volatile. It is therefore conceivable that vapor extraction activities or molecular diffusion of the VOCs through the vadose zone could be a source to the local atmosphere both before and during remediation activities. This potential is discussed in Section 5.4.4.

5.4.2 Groundwater Pathway

Groundwater monitoring at the Person Generating Station site indicates that the existing VOC plume in the upper flow zone of the aquifer is approximately 36 acres in areal extent with a maximum depth of 20 feet below the water table. Previous well surveys revealed that groundwater is not withdrawn for potable applications within a one-mile radius downgradient of the site (METRIC, 1992). However, one irrigation well is located to the northeast and approximately downgradient of the Person Generating Station site. This well, located almost one mile in the northeastern direction from the Person Generating Station site, meets the agricultural demands of the University of New Mexico championship golf course. This well draws from the deeper portions of the aquifer as it is screened in intervals from 200 feet to 992 feet below the ground surface (METRIC, 1992).

Recent sampling of this well did not detect any of the contaminants of concern. (Gary Richardson, METRIC, pers. comm.). Further, model data suggest that the shallow groundwater plume under all alternatives considered will not migrate significantly in this downgradient direction. Thus, it is unlikely that the VOC contaminants will travel to or impact this well.

Groundwater from the Person Generating Station plume does not appear to discharge into any of the receiving surface water bodies in the Rio Grande Basin area; rather, these sources act as perennial recharge areas for the upper aquifer.

The probability of future shallow groundwater wells in the plume area is very low, given both the predicted size and location of the VOC plume using a conservative modeling approach and the unlikelihood of the need to drill shallow wells in the area to meet either domestic or industrial water requirements. As discussed previously, significant concentrations are not expected to migrate significantly beyond the eastern right-of-way boundary of Interstate 25 under any of the remedial alternatives evaluated in this report. Therefore, the pathway of exposure to receptors from shallow groundwater is not currently complete and there are no risks.

5.4.3 Surface Water Pathway

There is no surface water contamination on the Person Generating Station site. Additionally, no complete pathway exists between contaminated soil and/or shallow groundwater and surface water in the Rio Grande Basin area.

5.4.4 Air Pathway

The installed closure cap over the source waste tank area should provide a surface seal preventing the direct volatilization of VOCs from the contaminated source area into the local atmosphere. However, molecular diffusion of VOCs from contaminated near-surface soil and soils previously covered by the receding groundwater table could introduce VOCs very slowly into the local atmosphere. Another potential air pathway may exist during remediation activities. Concentrations of VOCs emitted during soil vapor extraction activities may require treatment if atmospheric dispersion does not significantly reduce concentrations to which site workers could be exposed.

5.5 SUMMARY

In conclusion, initial modeling predicts that contaminated groundwater at the Person Generating Station site will have to be pumped and treated approximately 6 years using four extraction wells to achieve target remediation levels (e.g., MCLs). Although the model predicts that at least 6 years may be necessary to treat shallow VOC-contaminated groundwater to attain MCLs, it is possible that actual remediation activities could be less effective than predicted by this model. PCE is a relatively persistent compound and may adsorb strongly to surrounding soils requiring a longer pumping period. The model was based on a number of site-specific assumptions that may result in either overestimating the degree and extent of contamination at the site under future scenarios or underestimating the time required to implement treatment.

No current exposure pathways are completed for the Person Generating Station site. The model also predicts that no exposure pathways are likely to be completed in 20 years even under the most conservative model assumptions and existing site information. Given this low risk of exposure to contamination at the site, consideration should be given to implementing a limited pump-and-treat remediation program at the site with the goal of reducing contaminants to the lowest concentration that is technically feasible. Based on model predictions, the majority of the contamination

will be removed during the initial 3 to 5 years of pumping, with asymptotic levels reached after 6 to 9 years of pumping. When actual contaminant removal data indicates pumping has reached the level of diminishing returns, an additional analysis of the transport, fate and risks of remaining contaminants should be completed.

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APPENDIX A

AIR STRIPPER INFORMATION

Ejector Systems Incorporated

910 National Avenue, Addison, IL 60101-9812

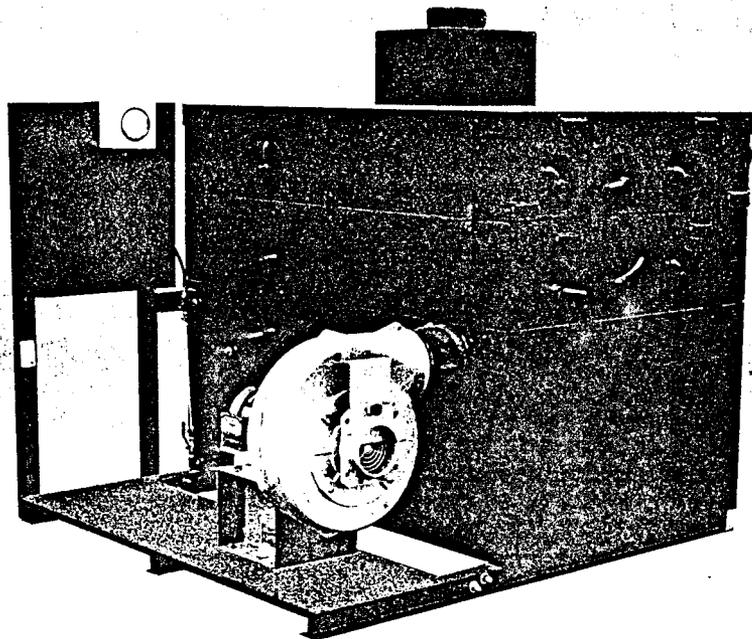
CASCADE LP500 SERIES LOW PROFILE AIR STRIPPER

Standard Specifications:

- Dimensions of 1 tray unit: 57" high x 65.5" wide 71.5" long
- 16 gauge epoxy-coated carbon steel trays
- Approximate weight of 1 tray unit = 1240 lbs.
- Volume per tray = 30 gallons
- 400 scfm AMCA Type B spark-resistant pressure blower
- Influent 3" FNTF
- Effluent 3" FNTF
- Effluent sump working volume = 98 gallons
- Blower back pressure gauge
- 8" clean-out port on sump
- Sight glass on sump

Options:

- NEMA 4 blower motor starter
- Explosion-proof blower motor starter
- TEFC transfer pump
- Explosion-proof blower
- Explosion-proof transfer pump
- Hi/low air switch
- High effluent switch



Patent Pending

708-543-2214

1-800-OIL-LEAK

Fa 708-543-2014

D.B. 6-7-93
LP500.DWG

LP500 SERIES CASCADE AIR STRIPPER

PATENT PENDING

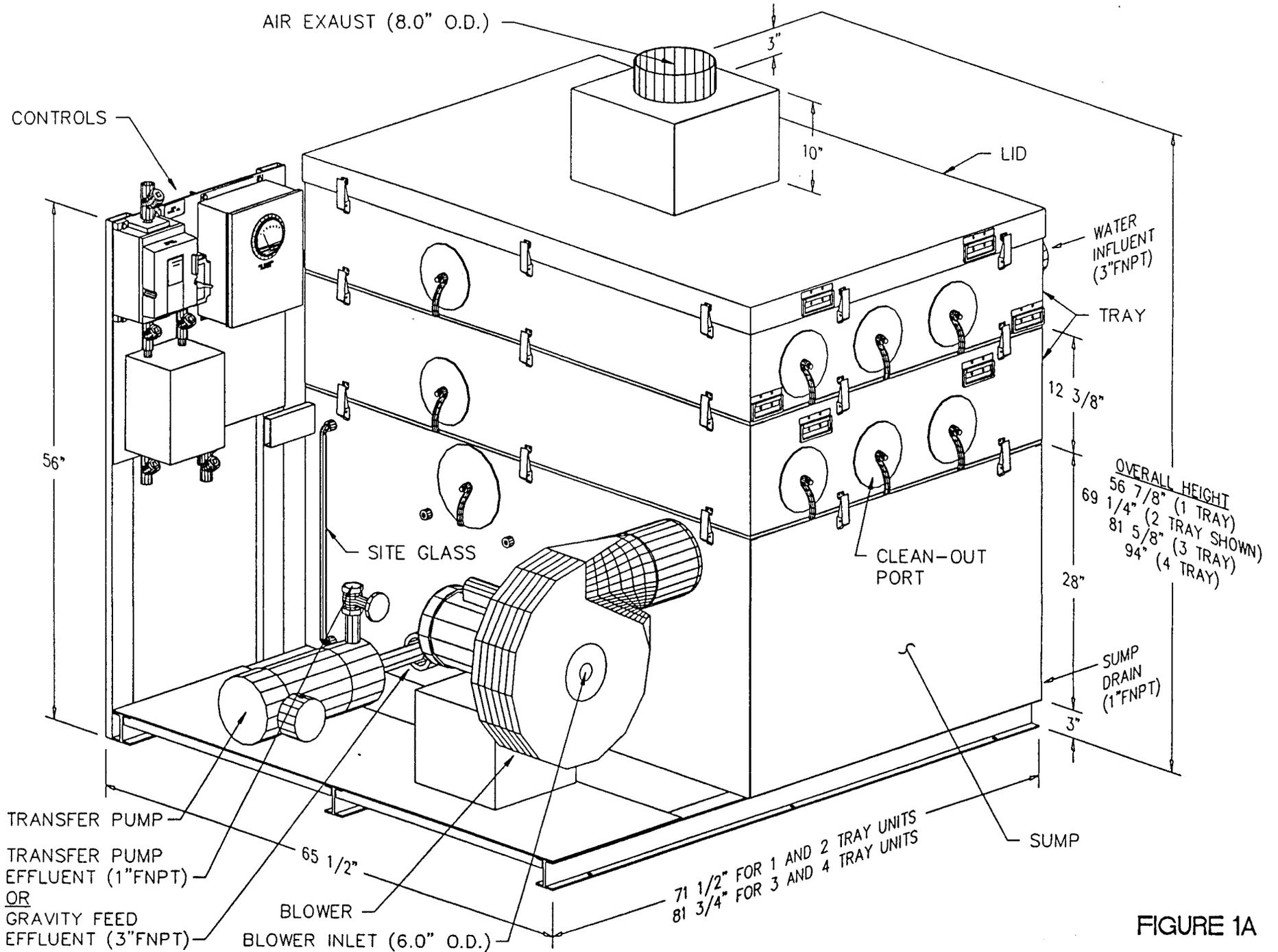


FIGURE 1A

LP500 SERIES CASCADE AIRSTRIPPER TRAY
(ISOMETRIC VIEW)

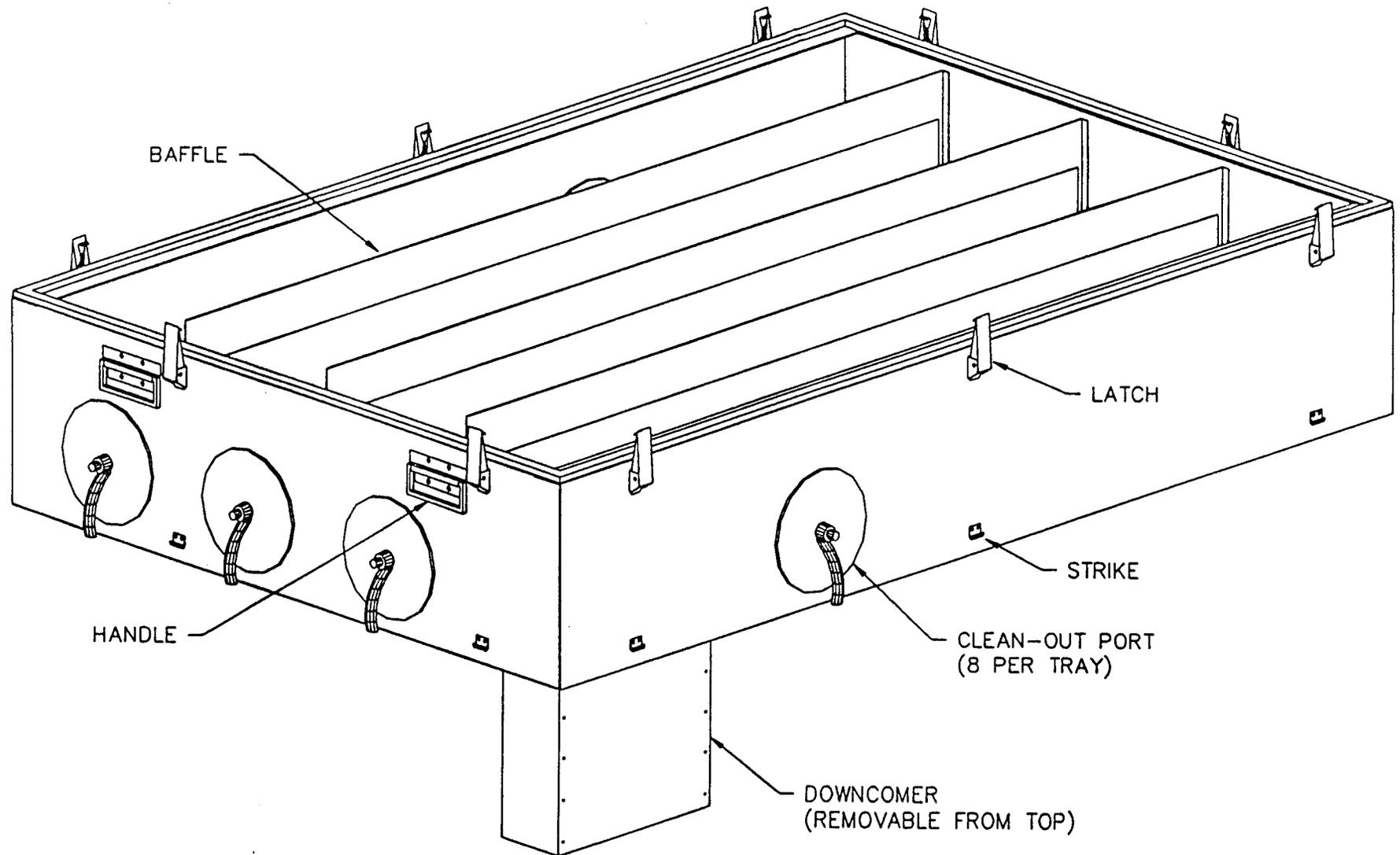


FIGURE 1.4A

LP500 SERIES CASCADE AIRSTRIPPER TRAY
(TOP VIEW)

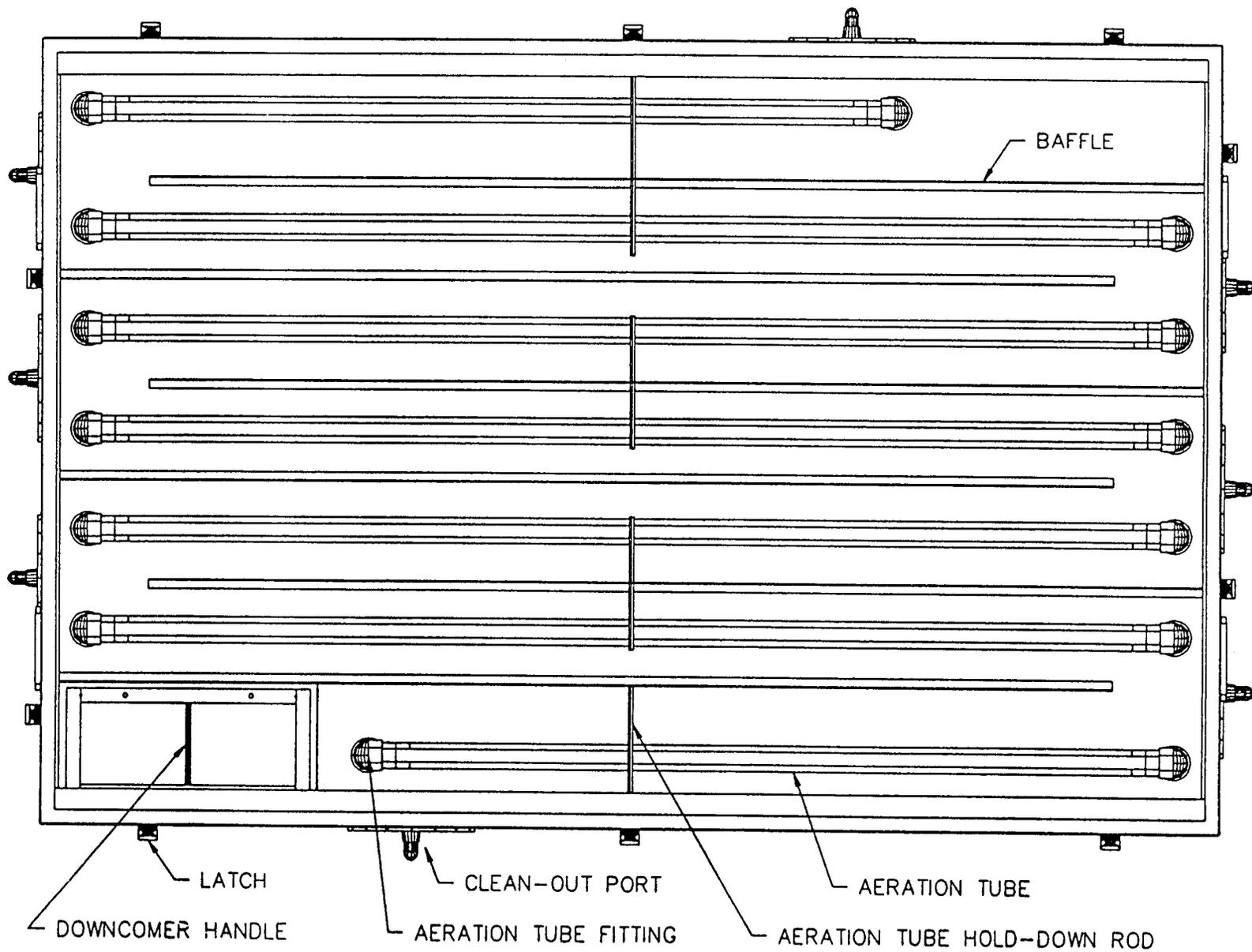


FIGURE 1.4B

SECTION 500 LOW PROFILE AIR STRIPPER

500.1 DESCRIPTION

500.11 Scope

Equipment to be furnished under this section includes the complete air stripping system as described hereinafter. The air stripping system shall be manufactured by Ejector Systems Inc.

Air stripping equipment which includes, but is not limited to, stripper sump base, lid, blower, air header, and trays. The equipment specified in this section shall be furnished by a single responsible supplier, that is established in the design and manufacture of air stripping systems.

The air stripping system furnished under this section of the specifications shall be placed in a well ventilated area that will not affect the performance of the system. All appurtenant components of the stripping system not furnished supplier, such as pipe and fittings for inlet and effluent water streams, pipe and fittings for effluent gas, concrete pads or foundations for system placement, and electrical controls and supplies not mentioned herein shall be furnished by others.

500.12 Functional Description

Volatile Organic Carbons (VOC's) will be air stripped through the mass transfer of hydrocarbons from the liquid phase to gaseous phase shall be done through the use of a combination of aeration tubes and baffles incorporated within a tray type system.

In conjunction with the aerator/baffle network, multi-media packing of any sort will not be allowable in any form within the system.

Flow into the top tray of the air stripper will be through a 3 inch NPT fitting. Flow through the stripping unit will be via gravity. Effluent from the stripper will be through a 3 inch NPT fitting located at the bottom of the stripping unit.

Within each tray, baffles will direct flow direction to an opening allowing contaminated water to flow to the next lower tray in the system.

Aeration tubes will constantly be supplied with air from both sides of the tube.

Gaseous hydrocarbon that has been stripped will exit the stripping unit via an 8 inch diameter adaptor at the top of the air stripper.

500.13 Design and Materials of Construction

A. Air Stripper Trays and baffles within each tray shall be constructed of black epoxy-coated steel. All trays and baffles shall be welded and the entire unit shall be tested for leaks prior to shipment.



Each tray shall measure 3.6 feet wide by 5.5 feet long. Each tray shall incorporate 6 baffles to make 7 water legs to direct water flow.

B. Aeration Tubes shall be constructed of 0.035 inch thick Nylon and shall be used to introduce air to contaminated groundwater. Each tube shall measure approximately 5 feet in length and have an outside diameter of 1.425 inches. Two rows of 5/16 inch diameter holes spaced 1-1/2 inches apart shall die cut into all aeration tubes. If a porous plate or sieve tray design is proposed in lieu of aeration tubes, the minimum hole diameter shall be 5/16 inch to inhibit plugging.

C. The base and Lid shall be constructed of carbon steel and the base shall be capable of supporting the full operating system. They shall be painted with an epoxy type coating. All system components shall rest on base including blower, blower motor, and transfer pump (if included).

D. The air stripper blower shall be of the radial-blade pressure type. Blower wheel shall be mounted directly on the motor shaft. All air stripper units with 2 or less trays shall be fitted with a 3 HP single-phase blower. All units with 3 or more trays will use a 5 HP single-phase blower. All units will require 220-volt single-phase power.

E. The system shall be field adaptable to a new air stripper size through the addition or subtraction of trays.

F. A transfer pump shall be furnished if effluent from the stripper cannot flow by gravity. The pump shall be a horizontal, close coupled centrifugal powered by a standard end mounted 3450 RPM ball bearing motor. It shall have a cast iron volute and cast iron, semi-open impeller. A mechanical seal shall be used where the shaft exits the volute. Liquid level control shall be through a float device with internal activation when it is tipped due to changing liquid level.

500.2 PERFORMANCE AND DESIGN REQUIREMENTS

500.21 Performance

A. Flow Rate and Influent Concentrations to Air Stripping System

1. Total flow = _____ GPM
2. Influent constituents (list all) and concentrations
3. Required effluent standards
4. Influent water temperature
5. Inorganic water chemistry such as iron, calcium carbonates.

B. The air stripping system shall be capable of reducing the influent constituents to desired effluent standards at the specified flow rate.

C. The following factors should not inhibit system operation nor quality of effluent stream.

1. High levels of suspended solids
2. High levels of iron
3. Build up of calcium carbonate deposits
4. High levels of water hardness
5. High levels of dissolved solids

500.22 Air Stripper Design

A. Type: Low-profile, tray with aerator and baffles.

B. Minimum number of trays: 1

C. Maximum number of trays: 4

D. Maximum allowable air stripper height: 94.5 inches

E. Maximum allowable air stripper base area including blower and all associated system equipment: 32.5 sq. ft.

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ENGINEERING SCIENCE, INC.
1700 Broadway, Suite 900
Denver, CO

80290

Attn: Doug Downey
Phone: 303-831-8100
FAX: 303-831-8208

Quote # 932096-00
Date: 10/14/93
Terms: Net 30 days
Freight: prepaid and added
FOB Addison
Quotation is valid for 60 days

Quote Specifications:

RE: PNM

Air Stripper design criteria: 50 gpm @ 50 F
Constituent Influent Design Effluent Projected Effluent

Constituent	Influent	Design Effluent	Projected Effluent
PCE	101 ppb	5 ppb	3 ppb
1,1 DCE	58 ppb	5 ppb	3 ppb
1,1,1 TCA	7 ppb	5 ppb	<1 ppb

AIR REQUIREMENT: ~400 CFM BLOWER: 3 HP TEFC 230 V/1 PHASE

We offer the following:

- 1 Cascade LP 5002 Air Stripper
2 tray air stripper with 3 HP blower motor
STANDARD AIR STRIPPER INCLUDES:
3" NPT gravity influent - no spray nozzle required.
Epoxy coated steel trays.
* 6" tray clean out ports (8 per tray).
* Removable nylon aeration tubes (7 per tray).
flexible wall, dual-5/16" hole pattern.
* Quick-release tray latches (10 per tray).
Lid with demister and 8" exhaust port.
* Flexible vent ducting connector.
Aluminum Blower
* AMCA Type B spark resistant
* Blower inlet guard with damper.
* Flexible inlet ducting connector.
Integral effluent sump base.
* 100 gallon working capacity.
* 8" clean out port.
* Removable sight glass with shut off valve.
Air pressure gauge.

OPTIONAL EQUIPMENT INCLUDED:

- NEMA 4 motor starter
- High sump level switch
- High/Low air supply switch

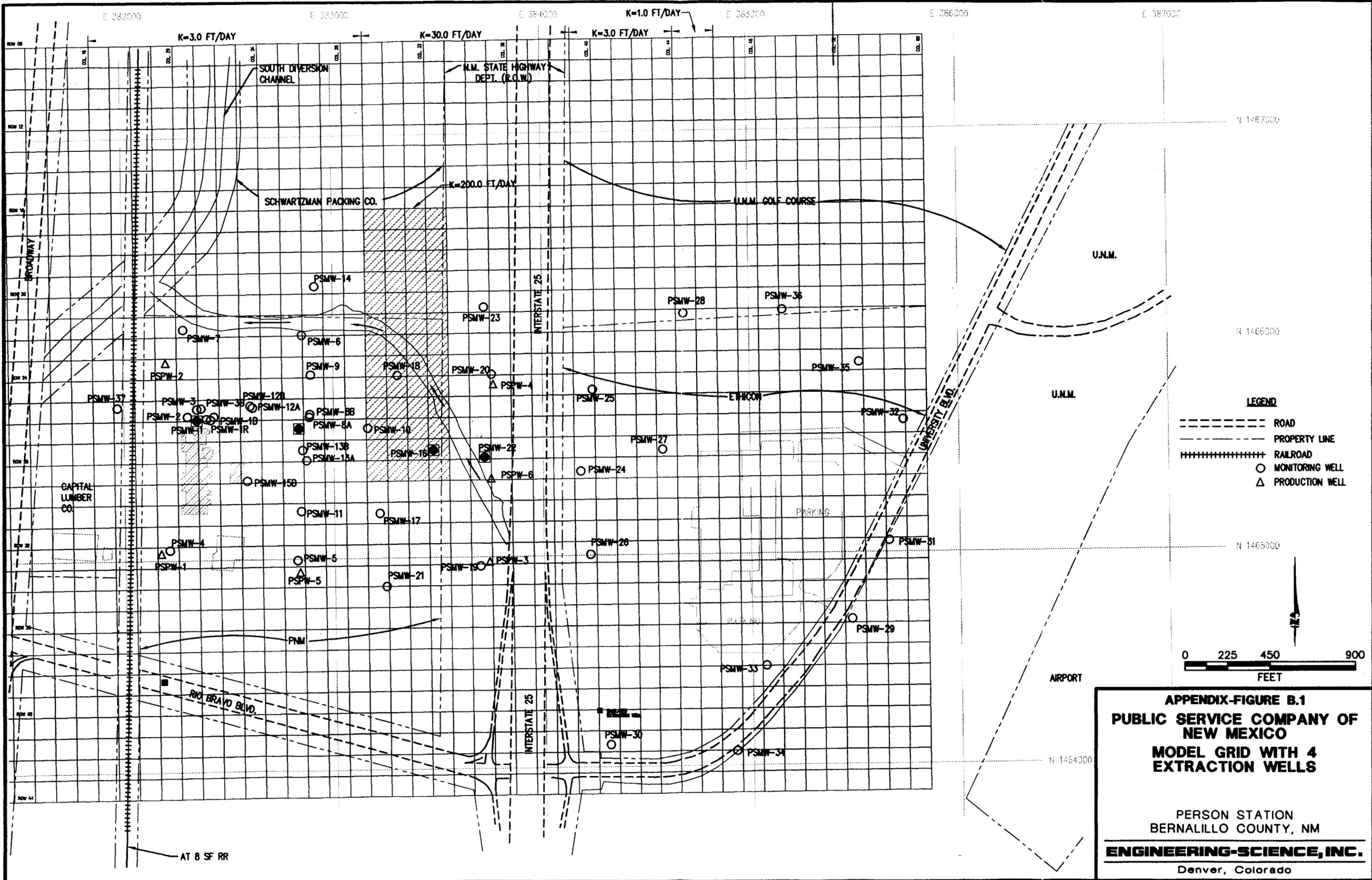
APPENDIX B
MODEL DEVELOPMENT AND CALIBRATION

The following plots identify key steps in the model development and calibration process. All model data and input parameters were developed from available site-specific information or were taken from the literature. The central portion of the model grid used is shown in Figure B.1. The model flow field under the external influence of four steady-state pumping wells is shown in Figure B.2. The calibrated model flow field is shown in Figure B.3. Additional information on MODFLOW from the user manual has been included for parties desiring further information on this groundwater model.

Note that a preliminary sensitivity analysis was conducted on the model with regard to vertical hydraulic conductivity. In the model information presented in Section 5, a value equal to 10 percent the horizontal hydraulic conductivity was used. To investigate the sensitivity of the model predictions to a change in the vertical hydraulic conductivity, a value equal to 1 percent the horizontal hydraulic conductivity was used. This order of magnitude change did not significantly affect the resulting plume under the influence of four extraction wells pumping for 6 years and 14 years of natural attenuation.

Increasing the retardation factor from 1.03 to 2.0 may increase the pumping time required to reduce contaminant concentrations within the shallow aquifer. Figure B.4 illustrates the potential effect of a larger retardation factor on the PCE plume following source removal, six years of pumping using 4 wells, and 14 years of natural attenuation.

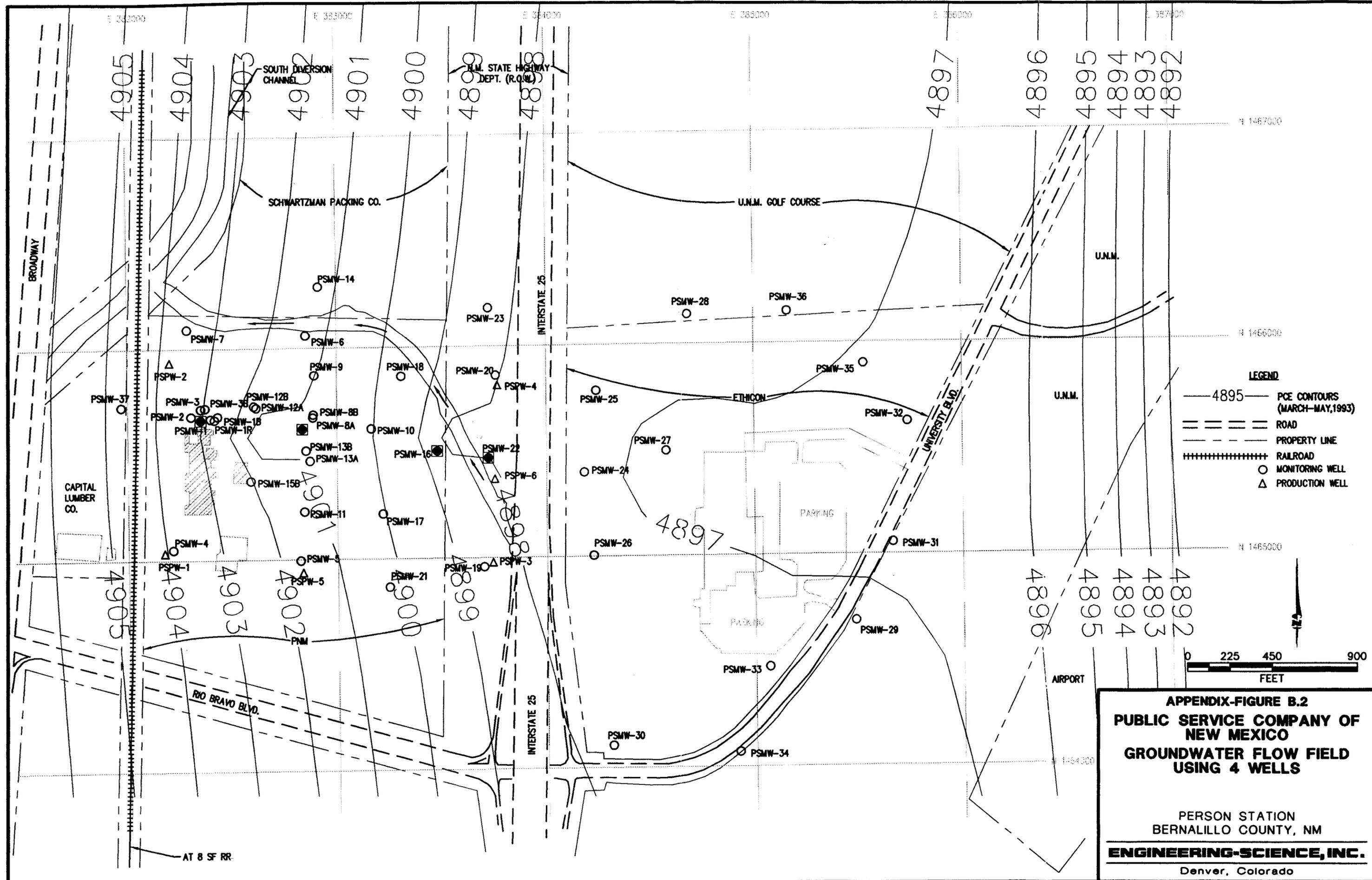
The model predictions focus on a discrete area of the entire model grid. Because little information is available for these areas outside of the existing monitoring well network, it is not possible to verify whether changes in these areas as required during the model simulations are reasonable. Further, calibration of the flow field using available potentiometric data and known hydraulic conductivity data was confined to single data sets. Potential seasonality changes and other effects not clearly reflected in the available site data are not included in the model simulations. Verification of the transport components in the MT3D model was also based on available site information. Values for dispersivity were analytically determined from historical site information.



APPENDIX-FIGURE B.1
PUBLIC SERVICE COMPANY OF
NEW MEXICO
MODEL GRID WITH 4
EXTRACTION WELLS

PERSON STATION
 BERNALILLO COUNTY, NM

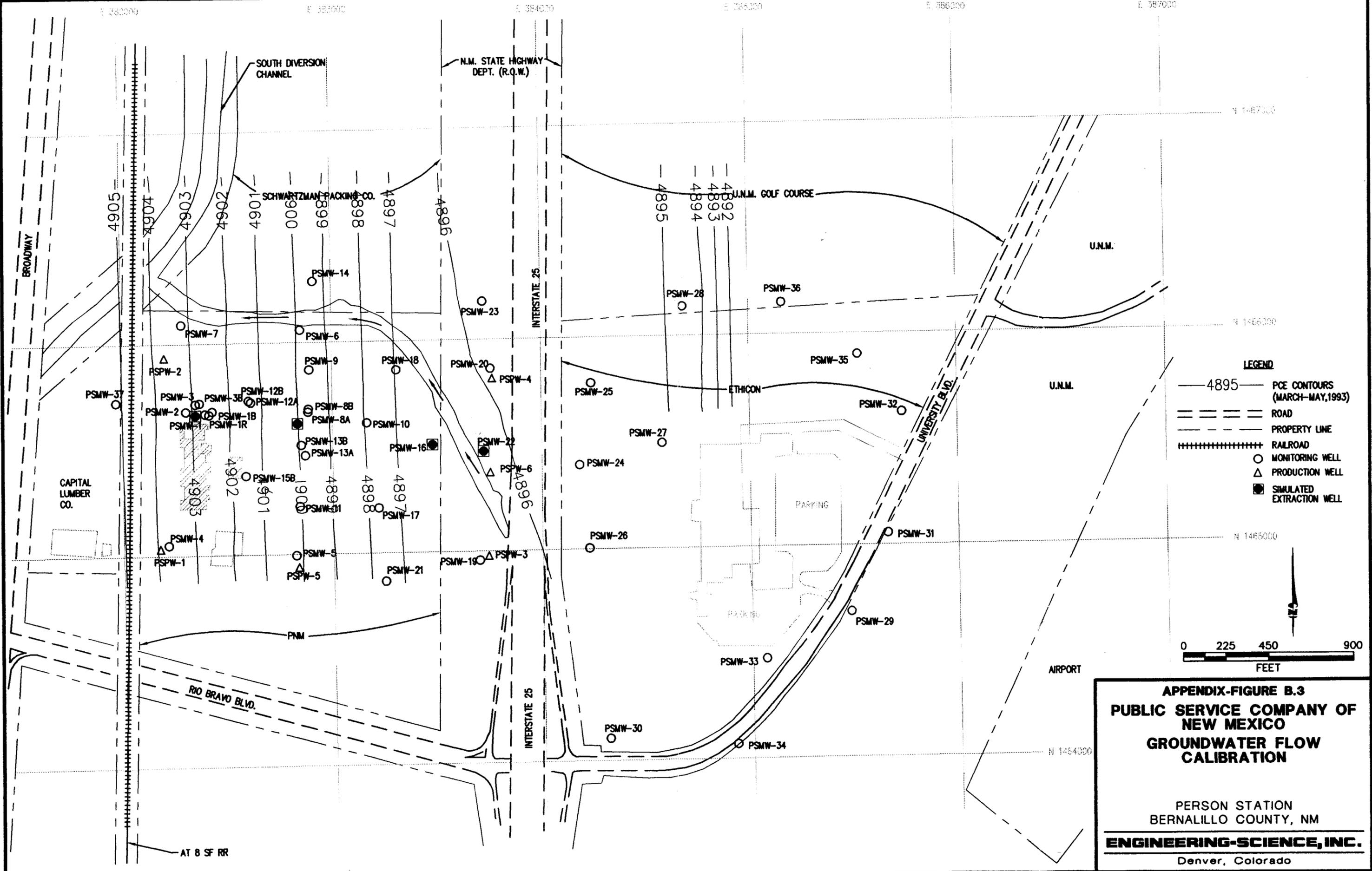
ENGINEERING-SCIENCE, INC.
 Denver, Colorado



APPENDIX-FIGURE B.2
PUBLIC SERVICE COMPANY OF
NEW MEXICO
GROUNDWATER FLOW FIELD
USING 4 WELLS

PERSON STATION
 BERNALILLO COUNTY, NM

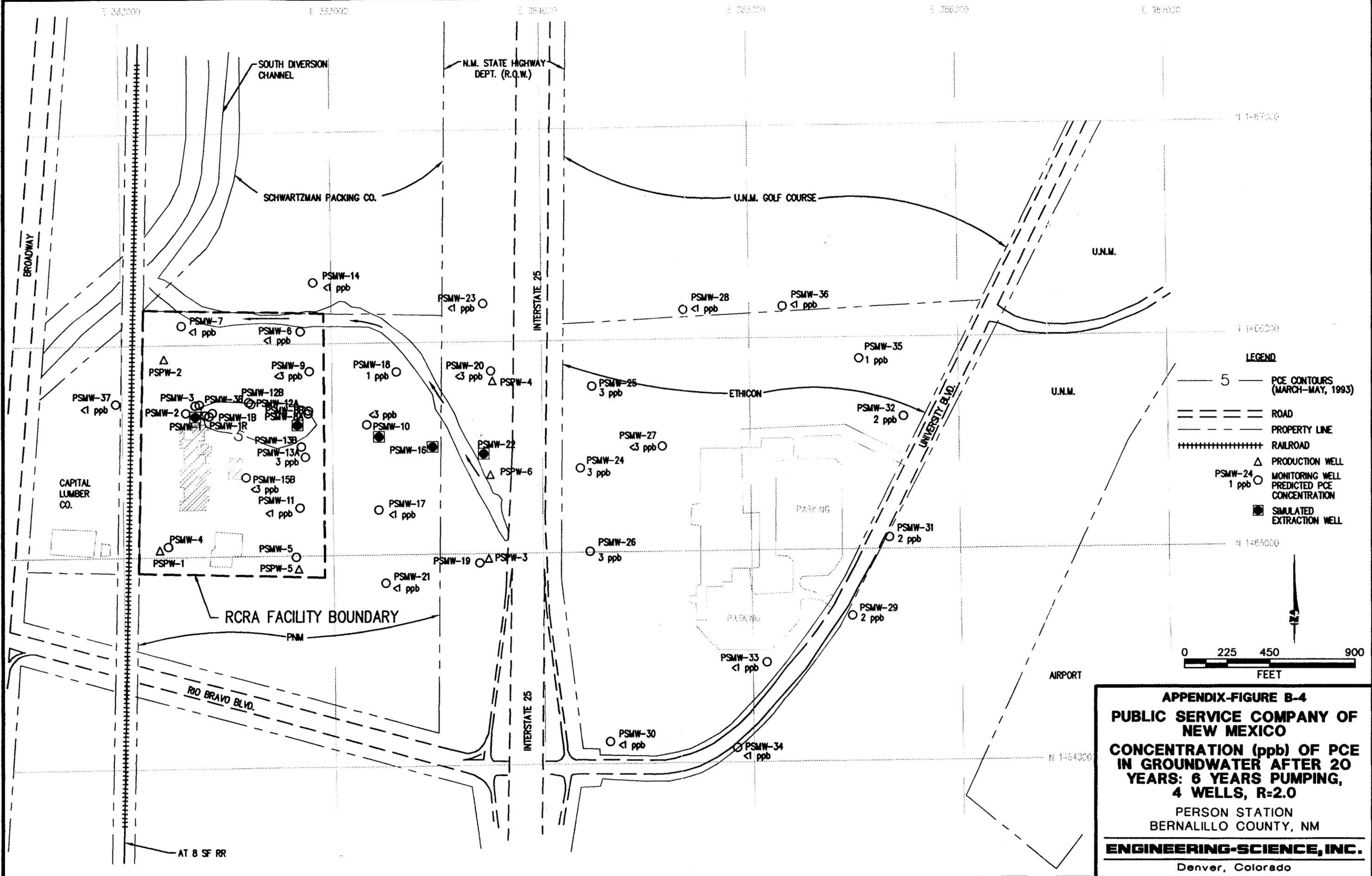
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 Denver, Colorado



APPENDIX-FIGURE B.3
PUBLIC SERVICE COMPANY OF
NEW MEXICO
GROUNDWATER FLOW
CALIBRATION

PERSON STATION
 BERNALILLO COUNTY, NM

ENGINEERING-SCIENCE, INC.
 Denver, Colorado



LEGEND

- 5 — PCE CONTOURS (MARCH-MAY, 1993)
- == ROAD
- - - - PROPERTY LINE
- ++++ RAILROAD
- △ PRODUCTION WELL
- PSMW-24 1 ppb MONITORING WELL PREDICTED PCE CONCENTRATION
- SIMULATED EXTRACTION WELL

0 225 450 900
FEET

APPENDIX-FIGURE B-4
PUBLIC SERVICE COMPANY OF NEW MEXICO
CONCENTRATION (ppb) OF PCE IN GROUNDWATER AFTER 20 YEARS: 6 YEARS PUMPING, 4 WELLS, R=2.0
 PERSON STATION
 BERNALILLO COUNTY, NM
ENGINEERING-SCIENCE, INC.
 Denver, Colorado

The USGS Modular Three-Dimensional Finite-Difference Ground-Water Flow Model, a.k.a. MODFLOW, by McDonald & Harbaugh (1988), has become one of the most popular and widely used models in the industry. One of the reasons for MODFLOW's widespread use is its modular nature, which allows users to add to the code without impairing the original function of the code.

In the last several years, the USGS has published several enhancements to MODFLOW that greatly increase the model's capabilities. Some of these enhancements are included in Geraghty & Miller's new release, MODFLOW386. Others will be supported by the next version of ModelCadTM, due out later this year. A partial listing of new USGS software and publications specifically related to MODFLOW is provided to keep the modeling community up-to-date with respect to this valuable model:

Hill, M.C., 1990, Preconditioned conjugate gradient 2 (PCG2), a computer program for solving ground-water flow equations, USGS WRIR 90-4048.

Kuiper, L.K., 1987, Computer program for solving ground-water flow equations by the preconditioned conjugate gradient method, USGS WRIR 87-4091.

Leake, S.A. and D.E. Prudic, 1988, Documentation of a computer program to simulate aquifer-system compaction using the modular finite-difference ground-water flow model, USGS OFR 88-482.

Pollock, D.W., 1989, Documentation of computer programs to compute and display pathlines using results from the USGS modular three-dimensional finite-difference ground-water flow model, USGS OFR 89-381.

Prudic, D.E., 1989, Documentation of a computer program to simulate stream-aquifer relations using a modular, finite-difference, ground-water flow model, USGS OFR 88-729.

Scott, J.C., 1990, A statistical processor for analyzing simulations made using the modular finite-difference ground-water flow model, USGS WRIR 89-4159.

We understand that other MODFLOW software is under review and development at the USGS. We will add to our list as these new features become available. If we have missed any MODFLOW enhancements, please let us know.

The USGS Modular Three-Dimensional Finite-Difference Ground-Water Flow Model, a.k.a. MODFLOW, by McDonald & Harbaugh (1988), has become one of the most popular and widely used models in the industry. One of the reasons for MODFLOW's widespread use is its modular nature, which allows users to add to the code without impairing the original function of the code.

Several MODFLOW enhancements were reported in the last Software Newsletter. Since that time, the USGS has published two new enhancements to MODFLOW. These new packages are listed below:

Harbaugh, A.W., 1990, A computer program for calculating subregional water budgets using results from the U.S. Geological Survey modular three-dimensional finite-difference ground-water flow model, USGS Open-File Report 90-392, 46 p.

Harbaugh, A.W., 1990, A simple contouring program for gridded data, USGS Open-File Report 90-144, 37 p.

A MODULAR THREE-DIMENSIONAL FINITE-DIFFERENCE GROUND-WATER FLOW MODEL

By Michael G. McDonald and Arlen W. Harbaugh

ABSTRACT

This report presents a finite-difference model and its associated modular computer program. The model simulates flow in three dimensions. The report includes detailed explanations of physical and mathematical concepts on which the model is based and an explanation of how those concepts are incorporated in the modular structure of the computer program. The modular structure consists of a Main Program and a series of highly independent subroutines called "modules." The modules are grouped into "packages." Each package deals with a specific feature of the hydrologic system which is to be simulated, such as flow from rivers or flow into drains, or with a specific method of solving linear equations which describe the flow system, such as the Strongly Implicit Procedure or Slice-Successive Overrelaxation.

The division of the program into modules permits the user to examine specific hydrologic features of the model independently. This also facilitates development of additional capabilities because new packages can be added to the program without modifying the existing packages. The input and output systems of the computer program are also designed to permit maximum flexibility.

Ground-water flow within the aquifer is simulated using a block-centered finite-difference approach. Layers can be simulated as confined, unconfined, or a combination of confined and unconfined. Flow associated with external stresses, such as wells, areal recharge, evapotranspiration, drains, and streams, can also be simulated. The finite-difference equations can be solved using either the Strongly Implicit Procedure or Slice-Successive Overrelaxation.

The program is written in FORTRAN 77 and will run without modification on most computers that have a FORTRAN 77 compiler. For each program module, this report includes a narrative description, a flow chart, a list of variables, and a module listing.

CHAPTER 1
INTRODUCTION

Purpose

Since their inception, the two- and three-dimensional finite-difference models described by Trescott (1975), Trescott and Larson (1976), and Trescott, Pinder, and Larson (1976) have been used extensively by the U.S. Geological Survey and others for the computer simulation of ground-water flow. The basic concepts embodied in those models have been incorporated in the model presented here. The primary objectives in designing a new ground-water flow model were to produce a program that could be readily modified, was simple to use and maintain, could be executed on a variety of computers with minimal changes, and was relatively efficient with respect to computer memory and execution time.

The model program documented in this report uses a modular structure wherein similar program functions are grouped together, and specific computational and hydrologic options are constructed in such a manner that each option is independent of other options. Because of this structure, new options can be added without the necessity of changing existing subroutines. In addition, subroutines pertaining to options that are not being used can be deleted, thereby reducing the size of the program. The model may be used for either two- or three-dimensional applications. Input procedures have been generalized so that each type of model input data may be stored and read from separate external files. Variable formatting allows input data arrays to be read in any format without modification to the program. The type of output that is available has also been generalized so that the user may select various model output options to suit a particular

need. The program was originally written using FORTRAN 66 (McDonald and Harbaugh, 1984). It has subsequently been modified to use FORTRAN 77. This report documents the FORTRAN 77 version. The program is highly portable; it will run, without modification, on most computers. On some computers, minor modification may be necessary or desirable. A discussion about program portability is contained in Appendix A.

The major options that are presently available include procedures to simulate the effects of wells, recharge, rivers, drains, evapotranspiration, and "general-head boundaries". The solution algorithms available include two iteration techniques, the Strongly Implicit Procedure (SIP) and the Slice-Successive Overrelaxation method (SSOR).

Organization of This Report

The purpose of this report is to describe the mathematical concepts used in this program, the design of the program, and the input needed to use the program. The program has been divided into a main program and a series of highly independent subroutines called modules. The modules, in turn, have been grouped into "packages." A package is a group of modules that deals with a single aspect of the simulation. For example, the Well Package simulates the effect of wells, the River Package simulates the effect of rivers, and the SIP Package solves a system of equations using the Strongly Implicit Procedure. Many of the packages represent options which the user may or may not have occasion to use. Each of the packages is described in a separate chapter of this report. Two preliminary chapters

describe topics relating to the overall program; Chapter 2 derives the finite-difference equation that is used in the model and Chapter 3 describes the overall design of the program. Chapter 14 describes utility modules that are used by various packages to perform special tasks. Appendices A-E cover topics relating to the operation of the model.

Chapters 4 through 13 describe individual packages. The description of each package consists of (1) a section entitled "Conceptualization and Implementation," (2) input instructions for the package, and (3) documentation of the individual modules contained in the package. The Conceptualization and Implementation section describes the physical and mathematical concepts used to build the package. For example, in the chapter describing the River Package, an equation is derived which approximates flow through a riverbed, and a discussion is provided to show how that equation can be incorporated into the finite-difference equation. Chapters 12 and 13 describe the solution procedures currently available in the model.

The input instructions in Chapters 4 through 13 are presented in terms of input "items." An item of input may be a single record or a collection of similar records, or it may be an array or a collection of arrays. (In the model described herein, three-dimensional arrays are always read as a collection of two-dimensional arrays, one associated with each model layer.) The input section in each chapter presents a list of the input items associated with the package described in that chapter; the entries in this list are numbered, and generally consist of two lines (sometimes followed by a note or comment). For items which consist of a single record or a group of similar records, the first line in the entry gives the names of the fields comprising the records, while the second line shows the format of those fields, in standard FORTRAN notation. For an input item which consists of an array, the first

line of the entry gives the name of the array, while the second line gives the name of the utility module which reads the array. Further details concerning utility modules are provided in Chapter 14.

For most of the packages, the list of input items is subdivided into two major sections. One of these falls under the heading "FOR EACH SIMULATION" and includes all items for which only one entry is needed in each simulation; the other falls under the heading "FOR EACH STRESS PERIOD", and includes those items for which several entries may be needed in each simulation (for example, pumping rate, which may change with time during the period represented in a simulation). These major sections of the input list are further subdivided by headings which indicate the modules (subroutines) which read the item, or, in the case of an array, which call a utility subroutine to read the array. Input items that are printed entirely in capital letters are used as FORTRAN variables or arrays in the model program; input items which appear in mixed upper and lower case print are terms used in the instructions to describe the input fields or procedures, and do not appear in the model itself as FORTRAN variables. Chapter 4, which describes the Basic Package, includes two lists of input items; one of these describes input which is always required, while the other describes input associated with the optional "output control" section of the Basic Package.

An explanation of input fields is presented following the list of input items in Chapters 4 through 13. This explanation is followed in most cases by a sample input for the package under consideration. In Chapter 4, again, the input items associated with the output control option are treated separately; thus an independent explanation of fields and sample input are

provided for output control.

In each simulation, the user must designate which of the options of the program are to be utilized, and must indicate the file from which the input for each option is to be read. This is done through a one-dimensional array, IUNIT; the entries in this array are the unit numbers associated with the required files by the computer operating system. A location in the IUNIT array is given at the beginning of the input sections in Chapters 5 through 13, and at the beginning of the input discussion for "output control" in Chapter 4. If the option is to be utilized, the user must enter, in the designated IUNIT array location, the unit number of the file or channel through which input for the option is to be read; if the option is not required a zero is entered in this location. Further discussion of the IUNIT array is provided in Chapters 3 and 4.

Following the input section in Chapters 4 through 13, each chapter provides a documentation of the modules making up the associated package. This documentation consists of a list of the modules in the package, followed by detailed descriptions of each of the modules. The detailed description of a module generally contains four documents: (1) a narrative description of the module, (2) a flow chart of the module, (3) a FORTRAN listing of the module, and (4) a list of the variable names which are used in the module. For very simple modules, the flow chart is omitted. The narrative description is a numbered list of the functions performed by the module showing the order in which they are performed. The flow chart is a graphic equivalent of the narrative. The blocks in the flow chart are numbered with the same numbers used in the narrative so that the two documents can be cross referenced. An explanation of terms used in the flow chart is contained on the sheet

with the flow chart. The program listing contains comments with numbers corresponding to those used in the flow charts and the narratives. The fourth record of the listing contains a comment showing the time and day that the module was last modified. The list of variables shows the name, range, and definition of every variable used in the module. If the variable is used only in that module, its range is given as "Module"; if it is used in other modules of the package, but not outside the package, its range is given as "Package"; if it is used in the modules of more than one package, its range is given as "Global."

To summarize the organization of this report, Chapters 2 and 3, and the "Conceptualization and Implementation" section of Chapter 4, provide discussions relevant to the overall design and functioning of the program; the formulation of coefficients representing flow within the aquifer is discussed under "Conceptualization and Implementation" in Chapter 5; Chapters 6 through 11 provide discussions of particular external sources or sinks and their representation in the model; and Chapters 12 and 13 discuss the operation of particular solvers for the systems of finite difference equations generated in the model. Input instructions for each package are provided in the relevant chapter; a discussion of input for utility modules is provided in Chapter 14. The appendices provide a sample problem, abbreviated input instructions, and discussions of certain computer-related topics.

Acknowledgement

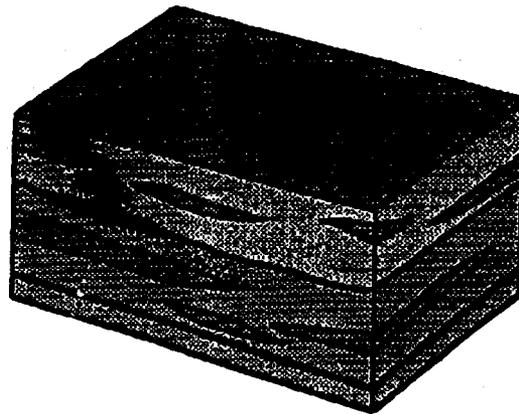
The authors wish to extend special thanks to Gordon Bennett. In addition to providing the administrative support for the model development, he provided encouragement and guidance along the way. His critical review of the report greatly improved its clarity.

STAES

M T 3 D

**A Modular Three-Dimensional Transport Model
for Simulation of Advection, Dispersion and Chemical Reactions
of Contaminants in Groundwater Systems**

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Abstract

mt3d: a modular three-dimensional transport model

This documentation describes the theory and application of a modular three-dimensional transport model for simulation of advection, dispersion and chemical reactions of dissolved constituents in groundwater systems. The model program, referred to as MT3D, uses a modular structure similar to that implemented in MODFLOW, the U. S. Geological Survey modular three-dimensional finite-difference groundwater flow model (McDonald and Harbaugh, 1988). This modular structure makes it possible to simulate advection, dispersion, sink/source mixing, and chemical reactions independently without reserving computer memory space for unused options. New transport processes and options can be added to the model readily without having to modify the existing code.

The MT3D transport model uses a mixed Eulerian-Lagrangian approach to the solution of the three-dimensional advective-dispersive-reactive equation, in three basic options: the method of characteristics (referred to as MOC), the modified method of characteristics (referred to as MMOC), and a hybrid of these two methods (referred to as HMOC). This approach combines the strength of the method of characteristics for eliminating numerical dispersion and the computational efficiency of the modified method of characteristics. The availability of both MOC and MMOC options, and their selective use based on an automatic adaptive procedure under the HMOC option, make MT3D uniquely suitable for a wide range of field problems.

The MT3D transport model is intended to be used in conjunction with any block-centered finite-difference flow model such as MODFLOW and is based on the assumption that changes in the concentration field will not affect the flow field measurably. This allows the user to construct and calibrate a flow model independently. MT3D retrieves the hydraulic heads and the various flow and sink/source terms saved by the flow model, automatically incorporating the specified hydrologic boundary conditions. Currently, MT3D accommodates the following spatial discretization capabilities and transport boundary conditions: (1) confined, unconfined or variably confined/unconfined aquifer layers; (2) inclined model layers and variable cell thickness within the same layer; (3) specified concentration or mass flux boundaries; and (4) the solute transport effects of external sources and sinks such as wells, drains, rivers, areal recharge and evapotranspiration.

Chapter 1

INTRODUCTION

1.1 PURPOSE AND SCOPE

Numerical modeling of contaminant transport, especially in three dimensions, is considerably more difficult than simulation of groundwater flow. Transport modeling not only is more vulnerable to numerical errors such as numerical dispersion and artificial oscillation, but also requires much more computer memory and execution time, making it impractical for many field applications, particularly in the micro-computer environment. There is obviously a need for a computer model that is virtually free of numerical dispersion and oscillation, simple to use and flexible for a variety of field conditions, and also efficient with respect to computer memory and execution time so that it can be run on most personal computers.

The new transport model documented in this report, referred to as MT3D, is a computer model for simulation of advection, dispersion and chemical reactions of contaminants in groundwater flow systems in either two or three dimensions. The model uses a mixed Eulerian-Lagrangian approach to the solution of the advective-dispersive-reactive equation, based on a combination of the method of characteristics and the modified method of characteristics. This approach combines the strength of the method of characteristics for eliminating numerical dispersion and the computational efficiency of the modified method of characteristics. The model program uses a modular structure similar to that implemented in the U.S. Geological Survey modular three-dimensional finite-difference groundwater flow model, referred to as MODFLOW, (McDonald and Harbaugh, 1988). The modular structure of the transport model makes it possible to simulate advection, dispersion, source/sink mixing, or chemical reactions independently without reserving computer memory space for unused options; new packages

involving other transport processes can be added to the model readily without having to modify the existing code.

The MT3D transport model was developed for use with any block-centered finite-difference flow model such as MODFLOW and is based on the assumption that changes in the concentration field will not affect the flow field significantly. After a flow model is developed and calibrated, the information needed by the transport model can be saved in disk files which are then retrieved by the transport model. Since most potential users of a transport model are likely to have been familiar with one or more flow models, MT3D provides an opportunity to simulate contaminant transport without having to learn a new flow model or to modify an existing flow model to fit the transport model. In addition, separate flow simulation and calibration outside the transport model result in substantial savings in computer memory. The model structure also saves execution time when many transport runs are required while the flow solution remains the same. Although this report describes only the use of MT3D in conjunction with MODFLOW, MT3D can be linked to any other block-centered finite-difference flow model in a simple and straightforward fashion.

The MT3D transport model can be used to simulate changes in concentration of single-species miscible contaminants in groundwater considering advection, dispersion and some simple chemical reactions, with various types of boundary conditions and external sources or sinks. The chemical reactions included in the model are equilibrium-controlled linear or non-linear sorption and first-order irreversible decay or biodegradation. More sophisticated chemical reactions can be added to the model without changing the existing code. Currently, MT3D accommodates the following spatial discretization capabilities and transport boundary conditions: (1) confined, unconfined or variably confined/unconfined aquifer layers; (2) inclined model layers and variable cell thickness within the same layer; (3) specified concentration or mass flux boundaries; and (4) the solute transport effects of external sources and sinks such as wells, drains, rivers, areal recharge and evapotranspiration.

1.2 SOLUTION TECHNIQUES

The advective-dispersive-reactive equation describes the transport of miscible contaminants in groundwater flow systems. Most numerical methods for solving the advective-dispersive-reactive equation can be classified as Eulerian, Lagrangian or mixed Eulerian-Lagrangian (Neuman 1984). In the Eulerian approach, the transport equation is solved with a fixed grid method such as the finite-difference or finite-element method. The Eulerian approach offers the advantage and convenience of a fixed grid, and handles dispersion/reaction dominated problems effectively. For advection-dominated problems which exist in many field conditions, however, an Eulerian method is susceptible to excessive numerical dispersion or oscillation, and limited by small grid spacing and time steps. In the Lagrangian approach, the transport equation is solved in either a deforming grid or deforming coordinate in a fixed grid. The Lagrangian approach provides an accurate and efficient solution to advection dominated problems with sharp concentration fronts. However, without a fixed grid or coordinate, a Lagrangian method can lead to numerical instability and computational difficulties in nonuniform media with multiple sinks/sources and complex boundary conditions (Yeh, 1990). The mixed Eulerian-Lagrangian approach attempts to combine the advantages of both the Eulerian and the Lagrangian approaches by solving the advection term with a Lagrangian method and the dispersion and reaction terms with an Eulerian method.

The numerical solution implemented in MT3D is a mixed Eulerian-Lagrangian method. The Lagrangian part of the method, used for solving the advection term, employs the forward-tracking method of characteristics (MOC), the backward-tracking modified method of characteristics (MMOC), or a hybrid of these two methods. The Eulerian part of the method, used for solving the dispersion and chemical reaction terms, utilizes a conventional block-centered finite-difference method.

The method of characteristics, which was implemented in the U.S. Geological Survey two-dimensional solute transport model (Konikow and Bredehoeft, 1978), has been used extensively in field studies. The MOC technique solves the advection term with a set of moving

particles, and virtually eliminates numerical dispersion for sharp front problems. One major drawback of this technique is that it needs to track a large number of moving particles, especially for three-dimensional simulations, consuming a large amount of both computer memory and execution time. The modified method of characteristics (MMOC) (e.g., Wheeler and Russell, 1983; Cheng et al., 1984) approximates the advection term by directly tracking the nodal points of a fixed grid backward in time, and by using interpolation techniques. The MMOC technique eliminates the need to track and maintain a large number of moving particles; therefore, it requires much less computer memory and generally is more efficient computationally than the MOC technique. The disadvantage of the MMOC technique is that it introduces some numerical dispersion when sharp concentration fronts are present. The hybrid MOC/MMOC technique (e.g., Neuman, 1984; Farmer, 1987) attempts to combine the strengths of the MOC and the MMOC techniques based on automatic adaptation of the solution process to the nature of the concentration field. The automatic adaptive procedure implemented in MT3D is conceptually similar to the one proposed by Neuman (1984). When sharp concentration fronts are present, the advection term is solved by the forward-tracking MOC technique through the use of moving particles dynamically distributed around each front. Away from such fronts, the advection term is solved by the MMOC technique with nodal points directly tracked backward in time. When a front dissipates due to dispersion and chemical reactions, the forward tracking stops automatically and the corresponding particles are removed.

The MT3D transport model uses an explicit version of the block-centered finite-difference method to solve the dispersion and chemical reaction terms. The limitation of an explicit scheme is that there is a certain stability criterion associated with it, so that the size of time steps cannot exceed a certain value. However, the use of an explicit scheme is justified by the fact that it saves a large amount of computer memory which would be required by a matrix solver used in an implicit scheme. In addition, for many advection-dominated problems, the size of transport steps is dictated by the advection process, so that the stability criterion associated with the explicit scheme for the dispersion and reaction processes is not a factor. It

should be noted that a solution package based on implicit schemes for solving dispersion and reactions could easily be developed and added to the model as an alternative solver for mainframes, more powerful personal computers, or workstations with less restrictive memory constraints.

1.3 ORGANIZATION OF THIS REPORT

This report covers the theoretical, numerical and application aspects of the MT3D transport model. Following this introduction, Chapter 2 gives a brief overview of the mathematical-physical basis and various functional relationships underlying the transport model. Chapter 3 explains the mixed Eulerian-Lagrangian solution schemes used in MT3D in more detail. Chapter 4 discusses implementational issues of the numerical method. Chapter 5 describes the structure and design of the MT3D model program, which has been divided into a main program and a number of packages, each of which deals with a single aspect of the transport simulation. Chapter 6 provides detailed model input instructions and discusses how to set up a simulation. Chapter 7 describes the example problems that were used to verify and test the MT3D program. The appendices include information on the computer memory requirements of the MT3D model and its interface with a flow model; printout of sample input and output files; explanation of several post-processing programs and tables of abbreviated input instructions.

1.4 ACKNOWLEDGEMENTS

I am deeply indebted to Dr. Charles Andrews, Mr. Gordon Bennett and Dr. Stavros Papadopoulos for their support and encouragement, and for reviewing the manuscript. I am also very grateful to Mr. Steve Larson and Mr. Daniel Feinstein, with whom I have had many helpful discussions. The funding for this documentation was provided, in part, by the United States Environmental Protection Agency.

APPENDIX C
LITERATURE ON TECHNOLOGY PERFORMANCE

Evaluating the Effectiveness of Ground Water Extraction Systems

by Jennifer L. Haley, Bill Hanson, Carl Enfield, and John Glass

Abstract

The most common process for remediating contaminated ground water is extraction and treatment. Data from 19 on-going and completed ground water extraction systems were collected and analyzed (U.S. EPA 1989b) to evaluate the effectiveness of this process in achieving cleanup concentration goals for ground water. This analysis indicated several trends including (1) containment of ground water plumes was usually achieved; (2) contaminant concentrations dropped significantly initially followed by a leveling out; (3) after the period of initial rapid decline, the continued decreases in concentration were usually slower than anticipated; and (4) certain data important to optimizing system design and operation had often not been collected during the site characterization phase.

Factors limiting the achievement of cleanup concentration goals fell into four basic categories: (1) hydrogeological factors, such as subsurface heterogeneity, low-permeability units, and presence of fractures; (2) contaminant-related factors, such as high sorption to soil and presence of non-aqueous phases (dissolution from a separate non-aqueous phase or partitioning of contaminants from the residual non-aqueous phase); (3) continued migration from source areas and the size of the plume itself; and (4) system design factors, such as pumping rates, screened intervals, and extraction well locations.

The findings of this study indicate that ground water extraction is an effective method for preventing additional migration of contaminant plumes and achieving risk reduction. However, the findings indicate that in many situations, it may not be practicable to rely solely on ground water extraction and treatment to achieve health-based cleanup concentrations throughout the contaminated zone and fulfill the primary goal of returning ground water to beneficial use.

This study suggests several recommendations (U.S. EPA 1989a) for improving ground water response actions including (1) actions to contain contaminant plumes should be initiated early; (2) data on vertical variation of hydraulic conductivity, distribution of the contaminant mass, and partitioning of contaminants to soil or a stationary phase in the saturated zone should generally be collected as part of the site characterization process; (3) remedial actions should be implemented in stages to better utilize information on aquifer response as the system is being designed and implemented; (4) remedial actions should be monitored and modified during operation to optimize system efficiency; and (5) methods to enhance extraction effectiveness and efficiency should be considered.

Introduction

Laboratory researchers looking at environmental processes and hydrogeologists involved in ground water contamination cleanup have been encountering several conditions that can limit the rate at which contaminants can be removed from the subsurface (U.S. EPA 1989c, Freeze 1989). The project described in this paper was initiated to assess the validity and prevalence of these limiting conditions in actual experiences with ground water extraction to date. The purpose of the project was to assess the effectiveness of ground water extraction systems in achieving specified goals at sites where ground water extraction systems had been operating for a long enough period of time to generate performance information.

Several sources of data were reviewed in an effort to identify operating ground water extraction systems and other systems where cleanup had been completed and pumping terminated. Information on 112 sites, including Superfund, RCRA, and industrial sites where

ground water response actions were being implemented by EPA, other federal agencies, states, or responsible parties, was collected and organized in a data base for review (U.S. EPA 1990b). Most of these sites, however, had not reached a full implementation phase and consequently were not useful for this study. Nineteen cases were identified as good candidates for more in-depth evaluation based on the data available on system effectiveness.

This paper presents the findings of the study and provides examples from the 19 case studies examined in detail (U.S. EPA 1990a) that illustrate the various factors that can affect the performance of ground water extraction systems. Finally, recommendations based on this study are summarized.

Background on Cases

The 19 case studies represent a variety of conditions frequently encountered when performing ground water extraction. Pertinent aspects of the 19 sites are provided in Table 1. Several general characteristics are presented

TABLE 1
Summary of Case Study Site Characteristics

Site No.	Site Name	Date of Initial Extraction	Remedial Objective	Chemicals Present	NAPLs Present	Geologic Environment	Innovative Technologies	Administrative Program
1	Amphenol Corp. New York	January 1987 ^a	Restoration ²	Organics	No	Unconsolidated glacio-fluvial sediments		RCRA
2	Black and Decker New York	May 1988 ^a	Restoration ¹	Organics	No	Glacial till and fractured sandstone	Fracture enhancement	RCRA
3	Des Moines TCE Iowa	December 1987 ^a	Restoration ¹	Organics	No	Unconsolidated glacio-fluvial sediments		Superfund
4	Dupont Mobile Plant Alabama	December 1985 ^a	Containment	Organics	No	Alluvial sand and clay		RCRA
5	Emerson Electric Co. Florida	December 1984 ^b	Restoration ¹	Organics	No	Sand		State Led
6	Fairchild Semiconductor California	1982 ^a	Containment	Organics	Maybe	Alluvial sand and gravel with silt, and clay layers	Slurry wall	State Led
7	General Mills Inc. Minnesota	Late 1985 ^a	Restoration ²	Organics	Maybe	Peat, glacial deposits, and fractured rock		State Led
8	GenRad Corp. Massachusetts	Late 1987 ^a	Restoration ¹	Low-sorption organics	No	Glacial sand, gravel	Intermittent pumping	RCRA
9	Harris Corp. Florida	April 1984 ^a	Wellhead treatment and restoration ¹	Organics	No	Sand and shell with a clay layer	Well points	Superfund and State Led
10	IBM Dayton New Jersey	March 1978 ^a	Was restoration, now containment	Organics	Yes	Sand with clay layers	Well points Reinjection	State Led
11	IBM San Jose California	May 1982 ^a	Restoration ¹	Organics	Yes	Alluvial sand and gravel with silt and clay layers		State Led
12	Nichols Engineering New Jersey	January 1988 ^a	Restoration ²	Organics	Maybe	Weathered and fractured shale		State Led
13	Olin Corp. Kentucky	1974 ^a	Containment	Organics	No	Unconsolidated glacio-fluvial sediments		State Led
14	Ponders Corner Washington	September 1984 ^a	Wellhead treatment	Low-sorption organics	No	Unconsolidated glacio-fluvial sediments	Vapor extraction	Superfund
15	Savannah River Plant South Carolina	September 1985 ^a	Mass reduction	Low-sorption organics	No	Coastal plain sand, silt, and clay layers		DOE
16	Site A Florida	August 1988 ^a	Restoration ¹	Organics	No	Limestone and sand		Superfund
17	Utah Power and Light Idaho	October 1985 ^a	Containment	Organics	Yes	Alluvium and fractured basalt	Intermittent pumping	RCRA
18	Verona Well Field Michigan	May 1984 ^a	Restoration ¹ and containment	Organics	Yes	Glacial sand, gravel, and clay	Vapor extraction	Superfund
19	Villie Mercier Quebec, Canada	1983 ^a	Containment	High- and low-sorption organics	Yes	Unconsolidated glacial sediments and fractured rock		Province of Quebec

^a Extraction still in progress.

^b Remediation completed and extraction system shut down in July 1987.

¹ Restoration to concentration goals equal to or less than health-based standards — MCLs or 10^{-6} excess cancer risk concentrations.

² Restoration to site-specific goals not directly related to health-based standards.

in the following text. In all cases, one of the goals of the extraction systems was to prevent additional migration of contaminants. Twelve of the cases also specified quantitative concentration or contaminant mass reduction goals, as well as containment.

The period of operation of the 19 extraction systems when available data were reviewed ranged from five months to six years. In most cases, the systems had already been operating longer than the projected time required for cleanup; however, concentration-based cleanup goals had not yet been attained and extraction was continuing.

The variety of contaminants encountered at these sites was limited. The primary contaminants in all but two cases were volatile organic compounds (VOCs). This is not surprising because VOCs are the most prevalent ground water contaminants found at Superfund sites and tend to be more mobile than other classes of compounds. Semivolatiles were present in two cases. Chromium, pesticides, and creosote were present at one site each.

The 19 case studies represent a broad spectrum of hydrogeologic settings from various geographic locations. Two of the sites are located in the northwestern

United States, seven are located in the Southeast, six in the Northeast, two in the Southwest, and two in the Midwest. This was considered useful in assessing the role that varying hydrogeologic and climatic conditions may play in impeding or promoting extraction of contaminants.

General Observations

Several trends were observed in the overall performance of the systems. As discussed previously, a common goal of all the actions was containment of the contaminant plume. In most of the 19 cases selected, this goal appeared to be successfully achieved. Ground water gradient data indicated that inward gradients toward the center of the plume were established with little or no movement of contaminants beyond the plume boundaries that existed at the initiation of the containment actions.

Contaminant mass removal was usually significant. Removal of thousands of pounds of contaminants (up to 130,000 pounds in one case) was not uncommon. However, the rate of mass removal often declined quickly. This initial drop in removal rate is thought to be the result of a combination of (1) removing ground water faster than the contaminants can desorb from the soil or diffuse from regions of low hydraulic conductivity to regions of higher hydraulic conductivity; (2) lowering water tables below the more contaminated soil; and (3) diluting concentrations by drawing in less contaminated ground water from surrounding areas. In some cases, this rapid decline may also reflect the removal of a slug of contaminated ground water, though there was no evidence of this in the cases evaluated.

Although concentrations in the ground water appeared to be reduced significantly, in all but one of the 19 cases, the levels remaining were generally above health-based standards for drinking water, which was the most common cleanup concentration goal of the actions. An example of the leveling out of contaminant mass removal rate is illustrated in Figure 1, taken from the Fairchild Semiconductor case study. The total mass of 1,1,1-Trichloroethane (TCA), isopropanol (IPA), acetone, and xylene removed from the subsurface from system startup in 1982 through May 1987 (for all wells in the system) was approximately 90,000 pounds. Figure 1 shows the change in contaminant mass over this time, with the typical leveling out of mass removal rate.

Factors Affecting Performance — Case Examples

The factors affecting the performance of the extraction systems examined in this study fell into the following four primary categories:

1. Aquifer properties, such as subsurface heterogeneity, and presence of low-permeability units or fractures
2. Contaminant properties, such as level of sorption to soil, of a separate non-aqueous phase, and partitioning to a separate non-aqueous phase
3. Adequacy of source removal and size of the plume itself

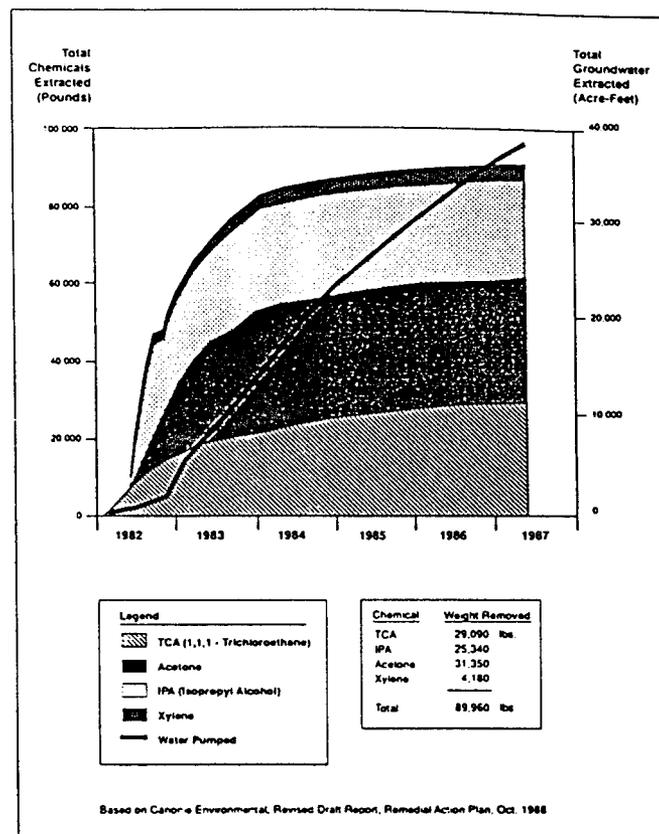


Figure 1. Cumulative totals of chemical mass and ground water volume extracted, 1982 to mid-1987, Fairchild Semiconductor Site.

4. System design, such as pumping rate, location of extraction wells, and depth/length of screened interval.

The following sections illustrate the impact these factors may have on the effectiveness of ground water extraction systems using examples from the case studies reviewed.

Aquifer Properties

All of the cases reviewed in this study reflected complications resulting from the heterogeneous nature of the subsurface. Homogenous hydrogeological systems below a contaminated site tend to be the exception rather than the norm. At a chemical plant site in Alabama, it appeared that the implications of the heterogeneous subsurface material were not accounted for in the design of the extraction system. The water-level data from monitoring wells located around the site indicated that hydraulic containment had been achieved, at least horizontally in the shallower, less permeable sediments. However, a rough contaminant mass balance on the system revealed that about half the contaminant mass was escaping the recovery wells. Though this was not a precise calculation, it did seem to indicate some unaccounted contaminant migration. A possible explanation for this apparent conflict is that contaminants were moving below the screened interval of the extraction wells. This explanation is supported by the fact that the hydraulic conductivity of the subsurface material increased with depth and all of the on-site wells were screened in the upper, less permeable portion of the

aquifer. In addition, a nearby production well screened at the lower depths continued to operate during this period and may have accelerated the vertical migration of contaminants.

The impact of low-permeability units in the subsurface is illustrated by the Ponder's Corner site in Lakewood, Washington. At this site, the variation of contaminant concentration with depth was assessed and correlated to the subsurface stratigraphy. This analysis indicated that almost 90 percent of the primary contaminant, tetrachloroethylene (PCE), was located in a low-permeability, silt and clay unit. Contaminant removal rates are limited not only by the slow rate at which ground water can be pulled through this unit, but also by the fact that the soil in this zone has a higher organic carbon content and consequently sorption of the PCE to the soil is enhanced.

Several of the case studies involved sites where fractures played a role in contaminant movement. At the Black and Decker site in Brockport, New York (Begor et al. 1989), the identification of discrete fractures led to the conclusion that recovery of trichloroethylene (TCE)-contaminated ground water would be difficult. In order to improve interconnection between the discrete fractures, explosives were set off in the bedrock creating a densely fractured zone that could be pumped to intercept the contaminant plume.

Contaminant Properties

Sorption is important in virtually all the case studies. The amount of contaminants sorbed to the soil is often not accounted for in estimating restoration time frames or in confirming that final cleanup goals have been attained. At the Savannah River Plant in Aiken, South Carolina, the contaminant mass in the ground water was estimated based solely on ground water concentrations. After three years of extraction, a comparison was made between the mass removed at the extraction wells and the difference in the estimated mass remaining in situ based on ground water concentrations before and after extraction. The mass actually removed by the system was 148,000 pounds; however, the ground water concentration comparison indicated that only 23,000 pounds had been removed. This discrepancy can be partly attributed to contaminants sorbed to the soil that were desorbing into the ground water as it was drawn to the extraction wells.

The presence of non-aqueous phase liquids that either float or sink in the aquifer can substantially increase the restoration time by acting as a continuing source of contaminants to the ground water. At the IBM Dayton Facility in South Brunswick, New Jersey, the extraction system was operated for six years, and concentrations appeared to be stabilizing at a level determined to be acceptable to the state. On-site extraction was then terminated. Continued ground water monitoring revealed that concentrations were increasing. It was concluded that this was the result of contaminant present in a non-aqueous phase more dense than water that had sunk within the aquifer. Because it would be very difficult to locate and completely remove the

"pockets" of entrapped non-aqueous liquids, the goal of the extraction system was changed to containment. Extraction was resumed at a lower pumping rate and will continue for an indefinite time period to ensure that off-site movement of the plume is prevented.

Problems can also result from non-aqueous phase liquids that are less dense than water. At the Verona Well Field site in Battle Creek, Michigan, a non-aqueous phase liquid layer approximately 1 foot thick was detected floating on the water table. Traditional product recovery techniques involving creation of a drawdown cone into which product would flow and could be recovered were used to reduce this layer to approximately 1 inch. At this point, product recovery techniques were no longer effective, but the remaining floating layer and residual saturation was sufficient to provide a source of contaminants to the ground water at levels above the cleanup goals established for the site. A vapor extraction system was then installed to remove the remaining product.

Adequacy of Source Removal

The ability of ground water extraction systems to achieve concentration reductions in the ground water may be hindered by the adequacy of measures taken to prevent continued contaminant migration from source areas. Soil cleanup levels are sometimes based on an evaluation of direct contact threats and may not account for the continued migration of contaminants to ground water. At an industrial site in Minnesota, concentrated wastes were removed from a disposal pit. Contaminated soil below the waste was not removed, despite sampling results that indicated significant levels of contaminants were present in the soil. Continued migration from the soils probably contributed to the difficulty experienced in efforts to reduce concentrations of the contaminants in ground water during extraction at this site.

System Design

Another factor affecting extraction performance is the design of the extraction system. In the case of the Alabama site previously discussed, the screened interval of the extraction wells may have been too shallow to contain the plume of contaminated ground water. At other sites, the locations of the wells, either clustered at the center of the plume or situated on the plume periphery, affected the rate at which the plume was drawn back. At the Fairchild Semiconductor site in San Jose, California, extraction wells were progressively shut off as the plume was drawn back. Not only does this reduce the volume of water that is pumped, but it allows for capture of contaminated ground water located at the edge of the capture zone and between the now-inactive extraction wells. This ground water would previously have been in "stagnation areas" where ground water does not flow in any direction due to the gradients created by the extraction systems.

Conclusions/Recommendations

The results of this evaluation highlight factors and

approaches that are prudent to consider in developing and implementing ground water response actions. The findings do not alter the primary goal of returning ground water to its beneficial uses in a reasonable time frame given the particular site circumstances (U.S. EPA 1990c). Rather, they argue for collection of data to allow for design of efficient cleanup systems and to more accurately estimate achievable cleanup levels and the time required for remediation. The recommendations cover three basic aspects of site remediation: consideration of early action, site characterization, and remedial action approach. In addition, it appears warranted to more routinely consider various methods to enhance the effectiveness of ground water extraction.

Recommendation 1: Plume Containment Should Be Considered Early

When ground water contamination is identified at a site, measures that can be implemented to prevent further migration of contaminants should be considered early in the site characterization phase. Implementing gradient control measures early can prevent the situation from getting worse and can provide valuable information on the plume response to pumping. Because the data needed to design a containment system can generally be collected relatively quickly, it will in many cases be valuable to prevent the contaminant plume from spreading while the site characterization and the selection and design of the full remediation system progresses. The determination made by the site manager regarding whether to implement a gradient control system would be based on existing information, best professional judgment, and data defining the approximate plume boundaries, contaminants present, and approximate concentrations. The benefits of initiating plume containment early should be weighed against the disruption that this will have on the existing gradient and possible loss of information on undisturbed contaminant migration patterns. The potential for spreading contamination from uncontrolled source areas (if containment wells are placed on the plume periphery) should also be considered. If it is determined that a gradient control system should be implemented, the advantages of initiating the action early should be maximized by carefully monitoring system response. In particular, ground water flow parameters should be monitored frequently (immediately before, during, and immediately after initiation of the action) to obtain information on system response.

Recommendation 2: Data that Will Assist in Assessing Contaminant Movement and Likely Response to Extraction Should Be Collected

In addition to water chemistry data required for the traditional plume characterization, assessments of contaminant movement and extraction effectiveness can be greatly enhanced by collecting more detailed information during construction of monitoring and extraction wells. Measurement of contaminant concentrations in the cores and stratigraphic variation of hydraulic conductivity are ways that may be used to gain this informa-

tion. Analysis of contaminant sorption to soil/sediment layers along with the hydraulic conductivity of the layers can provide a basis for estimating the time to reduce contaminant concentrations to specified levels and for identifying the presence of non-aqueous phase liquids. Cores taken from depths where relatively high concentrations of contaminants are identified might be analyzed to assess contaminant partitioning between the solid and aqueous phases. Measuring the organic carbon content in the soil can also provide useful information for estimating sorption in many cases (although total organic carbon measurements cannot be reliably determined where soil is heavily contaminated with organics).

Recommendation 3: Implement Ground Water Remediation Systems in a Staged Process

It will be beneficial at most sites to implement ground water remedial actions in stages. This might consist of installing one extraction well, initiating extraction, and observing the aquifer response as subsequent extraction wells are installed. This process will allow for siting extraction wells in optimum locations based on an evaluation of the area of influence of the initial well(s).

Recommendation 4: Allow for Modifications to the System Based on Information Gained During Its Operation

In many cases it may not be possible to determine the ultimate concentration reductions achievable in ground water until the ground water extraction system has been operated and monitored for some period of time. Remedial actions should be flexible and allow for modifications during operation. This iterative process of system operation, evaluation, and modification can effectively result in optimal system performance. If it appears that extraction is having a limited effect on portions of the plume, alternate, additional, or innovative measures may be considered. In some cases, it may not be practical to achieve the goal of returning the ground water to beneficial use (within an acceptable time frame) throughout the entire plume. Gradient control measures may be appropriate to prevent migration of contaminants from those areas.

Recommendation 5: Methods to Enhance Extraction Effectiveness and Efficiency Should Be Considered

It is clear from many of the case studies that variations made to system design and operation improved the effectiveness and efficiency of the extraction system. Some of these methods, such as infiltration/re-injection and construction of slurry walls, are fairly traditional. Others, like vapor extraction in conjunction with ground water extraction and fracture enhancement are relatively new and appear promising for certain types of situations. It may be appropriate to use some technologies, such as bioremediation, in a treatment train where extraction is used to achieve initial concentration reductions followed by the use of the more innovative technology to reduce concentrations further.

Finally, some alterations of traditional pumping systems may be worth consideration in most cases. This includes intermittent pumping to allow for contaminant and water level re-equilibration (U.S. EPA 1989c). Another worthwhile consideration is to determine how operation of the system (e.g., location of operating extraction wells) can be progressively modified based on observations of aquifer and plume response. Research at U. S. EPA laboratories is currently in progress to evaluate these and other approaches for addressing contaminated ground water and develop techniques to better assess contaminant migration patterns and processes.

Summary

Unless we identify effective, economical alternatives, ground water extraction will continue to be a primary method to reduce plume spreading and remove contaminants from ground water. An evaluation of several representative cases indicates that there are numerous factors and circumstances that can limit the effectiveness of ground water extraction as a remedial measure. These factors can often be recognized during site investigation by thorough data collection, which can serve to enhance the success of remedial actions. Also, remedies should be modified during system operation based on performance. In addition, it may be possible to implement a containment system prior to full site characterization to prevent contaminant migration as the investigation progresses, so long as the drawbacks of doing so do not outweigh the benefits.

Disclaimer

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Biographical Sketches

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Ground Water Remediation Using an Extraction, Treatment, and Recharge System

by Kurt O. Thomsen, Majid A. Chaudhry, Kostas Dovantzis, and Ronald R. Riesing

Abstract

Ground water remediation of volatile organic compound (VOC) contamination at a site in Michigan was initiated as a result of a consent agreement between the Michigan Department of Natural Resources (MDNR) and the responsible party. Under the direction of the MDNR, the responsible party conducted a remedial investigation/feasibility study using federal guidelines to define the extent of contamination at the site and to select a response action for site remediation. The selected alternative included a combination of ground water extraction, treatment, and recharge, and soil flushing. The extraction system withdraws ground water from various depths in heavily contaminated areas. The ground water is treated using an air stripper. A spray distribution system spreads effluent from the stripper over a recharge basin constructed over the most contaminated areas. Additional contaminant removal is achieved by volatilization from the spray and percolation through the gravel bed. Recharge water moves downward through the contaminated soils, thus flushing residual soil contaminants. The initial operating data demonstrated that the system can effectively remove trichloroethylene (TCE) from ground water (approximately 95 percent overall removal efficiency). The annualized capital and operation and maintenance (O & M) costs of the remedial action were estimated for several operating periods (15, 20, and 30 years).

Background

A refrigerator manufacturing facility is located on the lower end of two peninsulas formed by a reverse "S" meander of a river (Figure 1). In the late 1960s, local regulatory agencies granted the manufacturer permission to use the upper peninsula as a waste disposal area. Site contamination resulted from the waste disposal activities. Subsequently, a remedial investigation (RI) was conducted at the site to determine the nature and extent of contamination and to assess potential adverse effects to human health and the environment that could be associated with the site contamination.

The RI was conducted in three phases. The first phase was the exploratory phase, during which six well nests were installed. The results of this phase defined the general site stratigraphy. During the second phase of the RI, sampling was conducted to further define the site stratigraphy, hydrogeologic units, and extent of contamination. Trichloroethylene (TCE), tetrachloroethylene (PCE), and 1,1,1-trichloroethane (TCA) were the major contaminants found in the ground water. In addition, trace levels (<15 $\mu\text{g}/\text{L}$) of benzene, chloroform, 1,1-dichloroethane, 1,1-dichloroethylene, 1,2-cis-dichloroethylene, 1,2-trans-dichloroethylene, methylene chloride, toluene, and 1,1,2-trichloroethane were found in the ground water. Trichloroethylene (TCE) was the contaminant found in highest concentrations (as high as 35 mg/L) at the site.

Based on the results of the first and second phases, a consent judgment was negotiated between the state and manufacturer to remedy the contamination. The consent judgment required that ground water be treated until it consistently yielded a TCE concentration of no more than 15 $\mu\text{g}/\text{L}$. TCE was selected as the indicator parameter because, in most cases, its concentration in the ground water was an order of magnitude higher than either TCA or PCE. The 15 $\mu\text{g}/\text{L}$ cleanup criterion was based on the background concentration of TCE in the river adjacent to the site. The consent judgment also provided that the system was to be operated annually from April through October. As a result of the consent judgment, a third phase of the RI was conducted to better define the stratigraphy of the central portion of the site where most of the disposal activities took place (Figure 1). These data were needed to initiate a feasibility study of possible remedial alternatives.

The results of Phase 3 indicated that the complexity of site stratigraphy is responsible for variable ground water movement at the site. Ground water contamination is limited to an unconfined aquifer in communication with the river adjacent to the site. Discontinuous lenses of aquitard/aquiclude materials are generally present at two levels within the aquifer (Figure 2). The ground water contaminants are mostly confined to the saturated zone above the upper aquitard with significantly lower amounts

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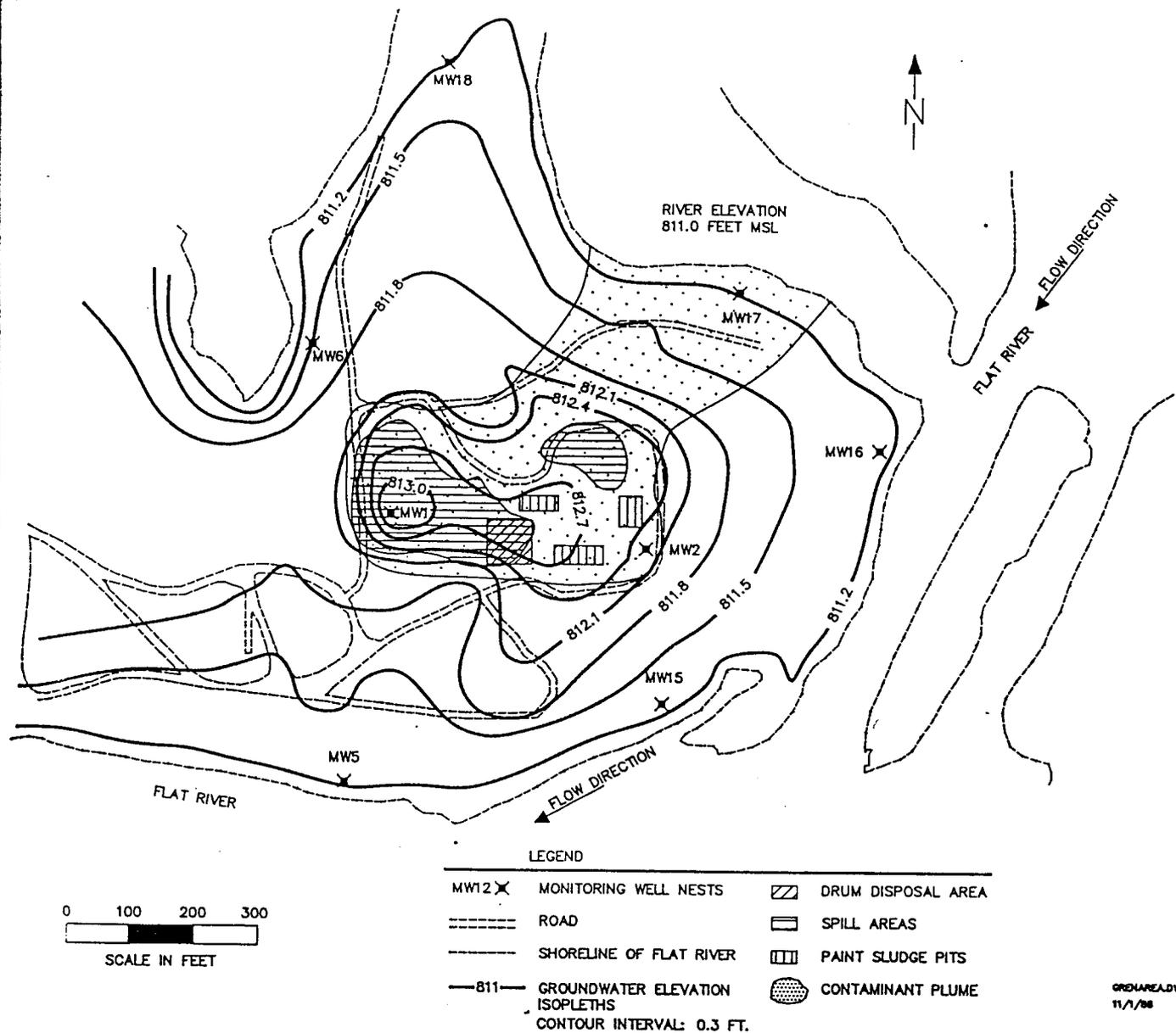


Figure 1. Site plot plan showing location of source areas, potentiometric surface and contaminant plume boundary.

of contaminants in the ground water between the two aquitard/aquiclude layers. No contamination was found between the lower aquitard/aquiclude layer and the confining layer underlying the aquifers. Ground water flows to the river (Figure 1) and vertically around and through the aquitard/aquiclude layers present at the site. Vertical ground water movement is upward to the river along the site boundary. However, at other locations both upward and downward ground water movement has been identified. Lateral ground water movement at the site is directed toward the river (Figure 1). Figure 1 also shows the boundary of the contaminant plume.

A feasibility study (FS) was conducted to identify, evaluate, and select a feasible alternative to remedy site contamination. The remedial alternatives considered in the evaluation included (1) no action (natural attenuation and/or dilution), (2) pump and treat using granular-activated carbon (GAC), and (3) pump and treat using air stripping. The selected remedial action for the site consisted

of ground water collection, treatment by air stripping, and discharge to a recharge basin. Pumping tests were conducted to obtain the information necessary to determine the number, location, and depth of ground water collection wells and the pumping rate needed to capture the ground water contaminant plume. Instead of discharging the treated ground water to the adjacent river, it was decided to discharge the treated ground water to a recharge basin in the area where most of the disposal took place (Figure 1), and thus flush the residual contaminants from the soils. After selecting the remedial action for the site, pilot testing was conducted. Pilot testing included pumping tests, treatability studies, and recharge tests. The results of the treatability studies showed an overall TCE removal efficiency of 93.9 percent (Thomsen, K.O. et al. 1986).

Pumping tests were conducted in five wells at three locations. Three pumping tests were conducted in the upper portion of the aquifer (above the upper aquitard/

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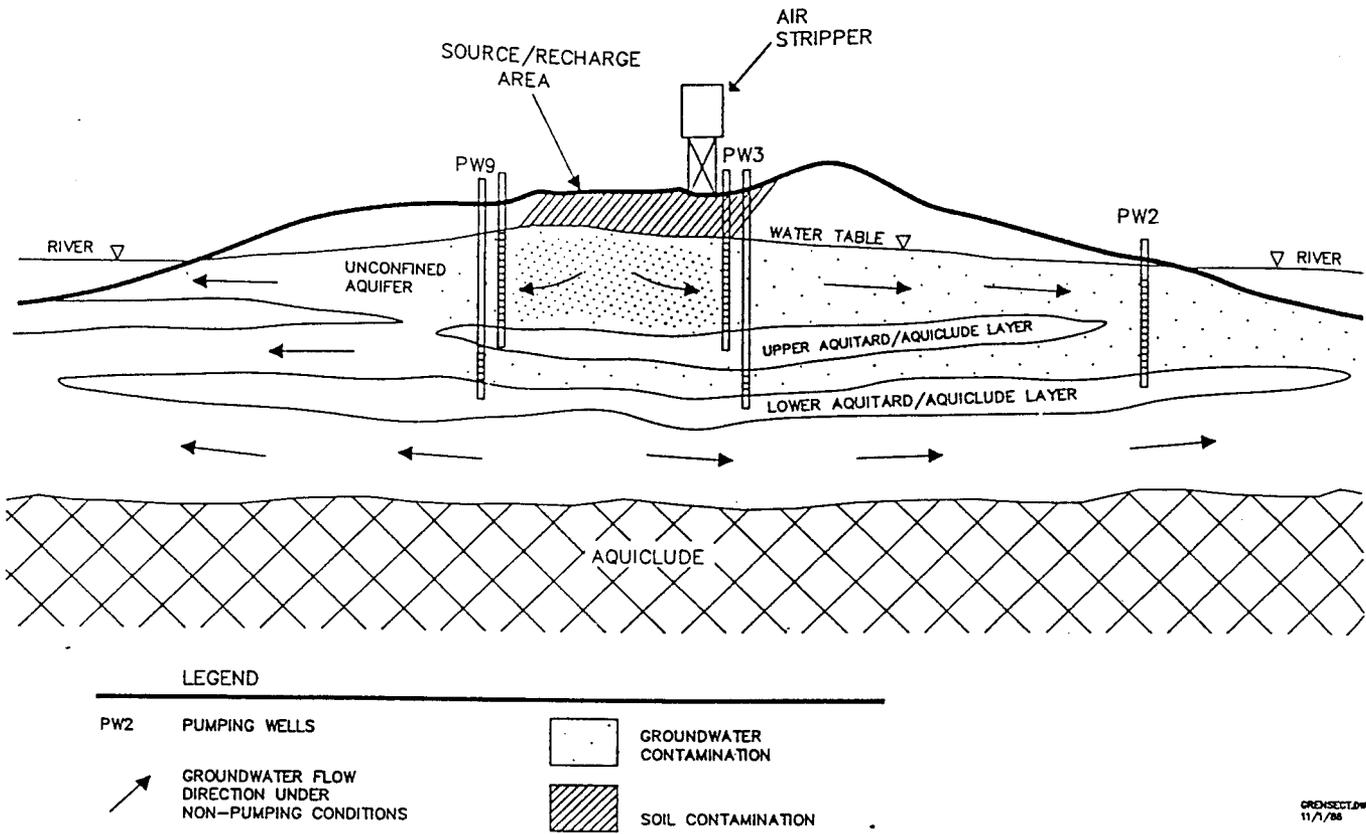


Figure 2. Schematic of a north-south cross section.

aquiclude layer), and two tests were conducted in the middle portion of the aquifer (between the aquitard/aquifer layers). Transmissivity ranged from 10,000 to 23,000 gallons per day per foot in both the upper and middle portions of the aquifer in the source area. One pumping test was conducted in a proposed plume interception well located near the river (PW-2). The transmissivity at this location was significantly higher (54,000 gallons per day per foot) than elsewhere.

Initially, it was thought that a 30- to 60-gallon-per-minute (gpm) treatment capacity would be adequate. Because air stripping was selected as the treatment process and the flow was expected to be low, the MDNR recommended using a draft-induced air stripping unit at the site. Treatability tests using this type of unit at a flow of 40 to 50 gpm resulted in a one-pass removal efficiency of 55 to 85 percent and a five-pass removal efficiency of 99.0 to 99.9 percent.

Because the treated ground water was to be used to flush the soil, a small spray distribution system was set up to determine the amount of additional volatilization that could be realized by distributing the treated water from the air stripper over the recharge area. The system was constructed using 2-inch Schedule 80 polyvinyl chloride (PVC) pipe and nylon nozzles having a 0.062-inch orifice. The nozzles were set into the top of the pipe at 5-foot intervals to direct the spray upward. The distribution system was placed on a 1- to 1.5-foot thick layer of pea gravel placed over a portion of the proposed recharge

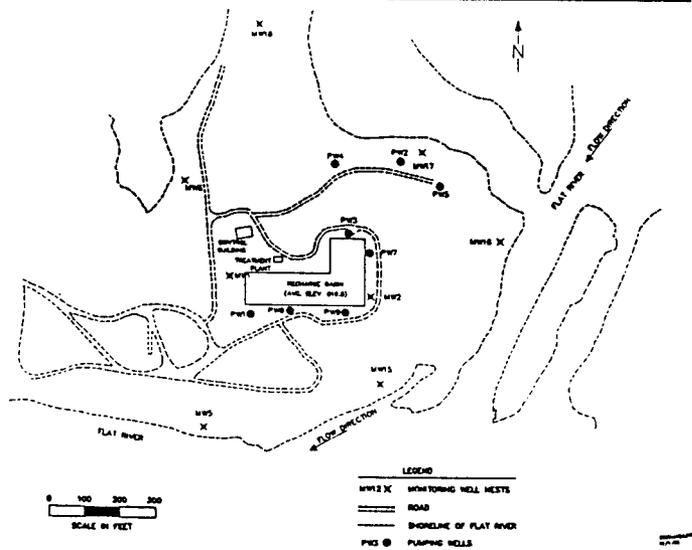


Figure 3. Site plot plan showing location of remediation system.

area. The gravel was meant to provide better distribution of the water, additional volatilization, and support for the distribution system. Tests showed an additional average removal of 78.7 percent by spray distribution and 15.2 percent by percolation through the gravel layer, yielding an average overall removal of 93.9 percent by the spray distribution system. Average removal efficiency for the combined pilot treatment and distribution system ranged from 97.3 to 99.1 percent.

The recharge capacity of the area to be used to recharge the treated water was estimated by conducting a series of

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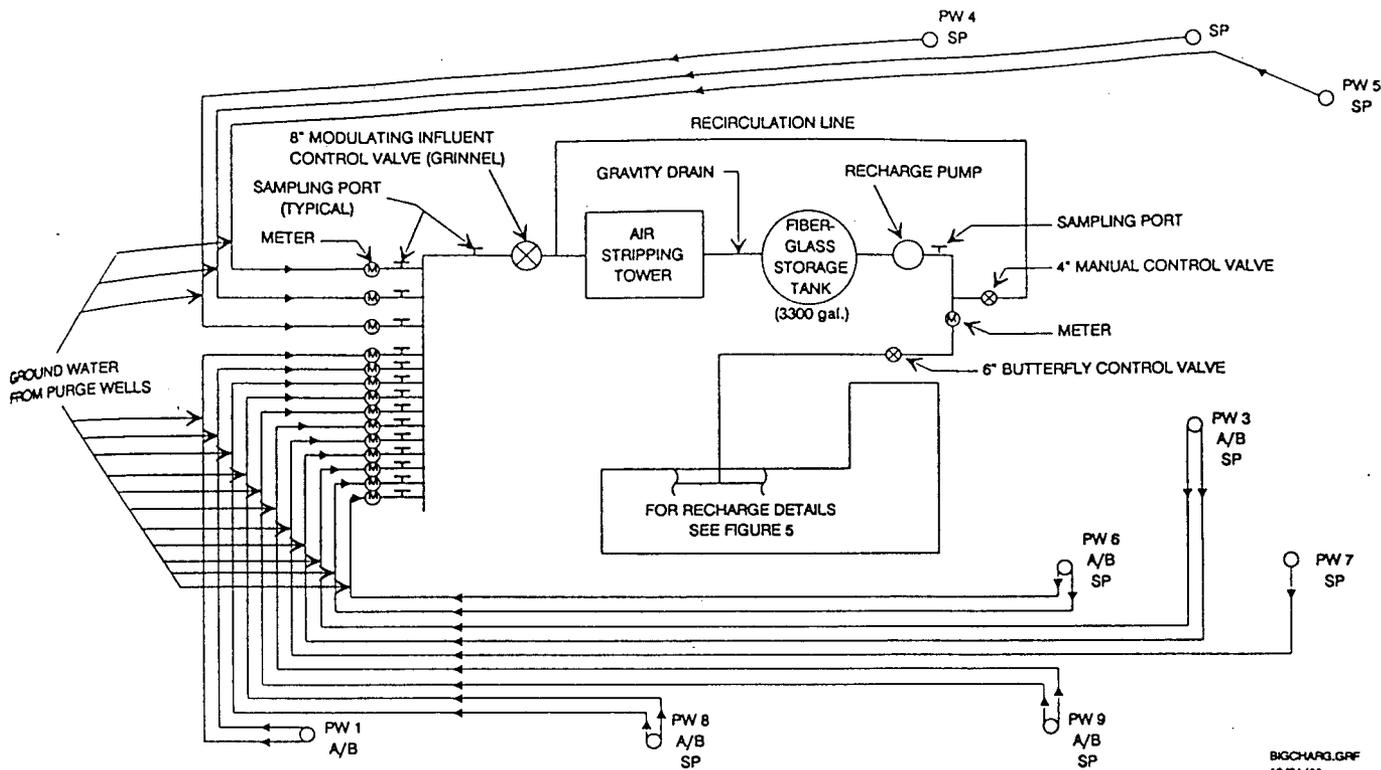


Figure 4. Process flow diagram.

infiltration tests over a 40,000-square-foot area. The recharge capacity of the proposed area was estimated at 800 gpm, but was specified at 600 gpm as a conservative measure to account for error involved in infiltration testing procedures.

Pilot testing was conducted concurrently with the third phase of the RI. The data collected during Phase III indicated that an effective remediation program would have to be significantly larger than originally envisioned. Therefore, the capacity of the remediation system had to be increased to approximately 600 gpm from the 30- to 60-gpm system proposed in the consent judgment.

The data obtained from the pumping tests, pilot treatability studies, and recharge tests were used to design and implement the remedial action for the site. The following sections present (1) the design and operating characteristics of the extraction, treatment, and recharge system; (2) a performance evaluation of the system based on initial operating data (July to October 1978); and (3) total capital costs and O & M costs. A site plot plan showing the location of the various components is presented in Figure 3.

Extraction System

The ground water extraction system consists of 14 purge wells at nine locations around the perimeter of the site. At five of these locations, a total of 10 wells pump from two levels within the upper glacial outwash aquifer (Figure 2); wells at the remaining four locations (four wells) pump from only one level. The maximum ground water withdrawal rate from the 14 wells is 600 gpm.

Shallow purge wells were drilled to the aquifer-aquitard layer interface, with the screen placed directly above the interface. A sump is located beneath the well screen to facilitate ground water flow toward the well. A

submersible pump located in the sump lifts ground water to the front end of the air stripping unit. Ground water from all purge wells is transported to the treatment system via 2- or 3-inch PVC pipes connected to an 8-inch header pipe. As a result, ground water from all 14 wells is combined before entering the air stripper. Each influent connection contains a strainer basket to filter out particulates, a flowmeter, a sampling port, a check valve, and a globe valve. Design and operating parameters of the collection system are presented in Table 1. A process flow diagram of the collection, treatment, and recharge system is presented in Figure 4.

Treatment System

The treatment system consists of a cooling tower used to air strip TCE from the contaminated ground water and a fiberglass storage tank used to store treated water prior to distribution. A cooling tower was used for several reasons: (1) a removal efficiency slightly better than the draft-induced air stripping unit, (2) a greater capacity, and (3) approximately half the cost of a packed tower air stripper having a comparable capacity. In the cooling tower, TCE is removed from the ground water with air that flows counter-current to the water flow. A fan located near the top of the tower draws air into the tower from two sides. Water is distributed over the tower and flows downward into the unit at an angle. Startup testing data collected between September and November 1986 showed an average TCE removal efficiency of 78 percent; this removal efficiency was obtained at an average ground water flow rate of 210 gpm and influent TCE concentration of 4000 $\mu\text{g/L}$.

The treatment system was designed to accommodate a maximum flow of 600 gpm. Once operation began, however, it was discovered that the recharge capacity of

the recharge area was approximately 210 gpm, one-third of the estimated capacity. Therefore, recharge capacity became the limiting factor in determining the actual treatment capacity.

A fiberglass storage tank, with an approximate holding capacity of 3300 gallons, is located underneath the stripping tower. Treated water is stored in this tank prior to distribution over the recharge area. To maintain the water at a preset level, the tank is equipped with a primary water level control that transmits a signal to a modulating influent control valve located on the 8-inch header pipe. The tank also has a backup unit that automatically interrupts power supply to the entire system, including purge well pumps, when the water in the storage tank reaches a predetermined high or low level. The treatment system is restarted manually after the cause of the shutdown is determined.

Flow from the storage tank can be periodically recirculated to the top of the treatment system for maintenance purposes, such as removing slime growth from the pipe lines. This is achieved via a 4-inch manual control valve and a 6-inch butterfly valve; opening the 4-inch valve allows flow to recirculate, while closing the 6-inch valve prevents flow from reaching the recharge area. To date, biological fouling has not been a problem, but deposits of precipitates caused by oxidation of minerals in the influent waste stream may be a future problem. Design and operating parameters of the treatment system are presented in Table 1.

Flow rate is measured before the water enters the header pipe and stripping tower (influent) and as the water is pumped from the storage tank to the recharge area. Water quality samples are collected at sampling ports located near the point where the influent and the recharge area flows are monitored and analyzed for the indicator parameter TCE.

Spray Recharge System

The treated ground water is distributed over a 40,000-square-foot recharge area located where the major disposal activities took place (Figure 1). The L-shaped area (Figure 3) is level and surrounded by a 3- to 4-foot berm providing a slope ranging from 3:1 to 2:1 around the perimeter to prevent runoff from the recharge area. The recharge area is subdivided into quadrants, each of which has similar piping and sample collection layouts. The purpose of this configuration is to provide the flexibility to direct and regulate the flow to areas based on the variation of recharge rates within the recharge area. A layer of pea gravel, 1 foot in thickness, covers the recharge area. This layer supports the piping network and acts as a drainage medium for water to percolate into the ground. Design parameters of the spray recharge system are presented in Table 1.

Ground water pumped from the storage tank to the recharge area is sprayed over the gravel layer through a PVC piping system. The piping system in each quadrant of the recharge area consists of two 4-inch header pipes located at each end of the quadrant and 10 lateral stringers at 10-foot intervals fitted with nozzles to provide upward

TABLE 1
Design and Operating Parameters
of Collection, Treatment, and Recharge System

Influent Flow Rate and Concentration:	
Maximum flow rate	600 gpm
Average flow rate	210 gpm
Average TCE concentration	4000 µg/L
Effluent TCE Goal:	
Pump, treat, and recharge until the TCE concentration in the ground water is 15 µg/L	
Collection System:	
No. of purge wells	14
Well diameter	4 to 6 inches
Well depth	20 to 30 feet
Submersible pumps	
Maximum flow rate	90 gpm
Total dynamic head	80 feet
Motor power	2 horsepower
PVC piping diameter	2 to 3 inches
Treatment System:	
Air Stripping Tower	
Length	17 feet
Width	6.5 feet
Height	9.5 feet
Maximum hydraulic loading rate	5.5 gpm/sq. ft.
Fan capacity	43,000 cfm
Fan motor power	10 horsepower
Fiberglass storage tank capacity	3300 gallons
Effluent discharge pump	
Maximum flow rate	330 gpm
Total dynamic head	65 feet
Motor power	10 horsepower
Spray Recharge System:	
Recharge area	40,000 feet ²
Berm slope, horizontal:	
vertical	2:1 to 3:1
Gravel bed depth	1 foot
PVC pipes with spray nozzles	
Diameter	2 inches
Pipe spacing	10 feet
Number of pipes	40
Nozzles	
Nozzle spacing	5 feet
Nozzles per pipe	13
Total number of nozzles	520
Nozzle capacity	0.60 gpm @ 40 psi
Orifice diameter	0.062 inches

(spray) water distribution. Aeration that takes place during spraying accounts for additional TCE removal. Water drains through the pea gravel layer and percolates into the ground. Once the water percolates into the ground, the extraction, treatment, and recharge cycle is complete.

Each quadrant (Figure 5) has a flow interceptor basin, located beneath the gravel layer, that collects some of the treated ground water before it percolates into the ground. Samples of treated water taken from these basins are used to monitor the overall TCE removal efficiency of the remediation system. For sampling purposes, each basin is connected to the surface with a flexible hose. Each time treated ground water samples are collected from these

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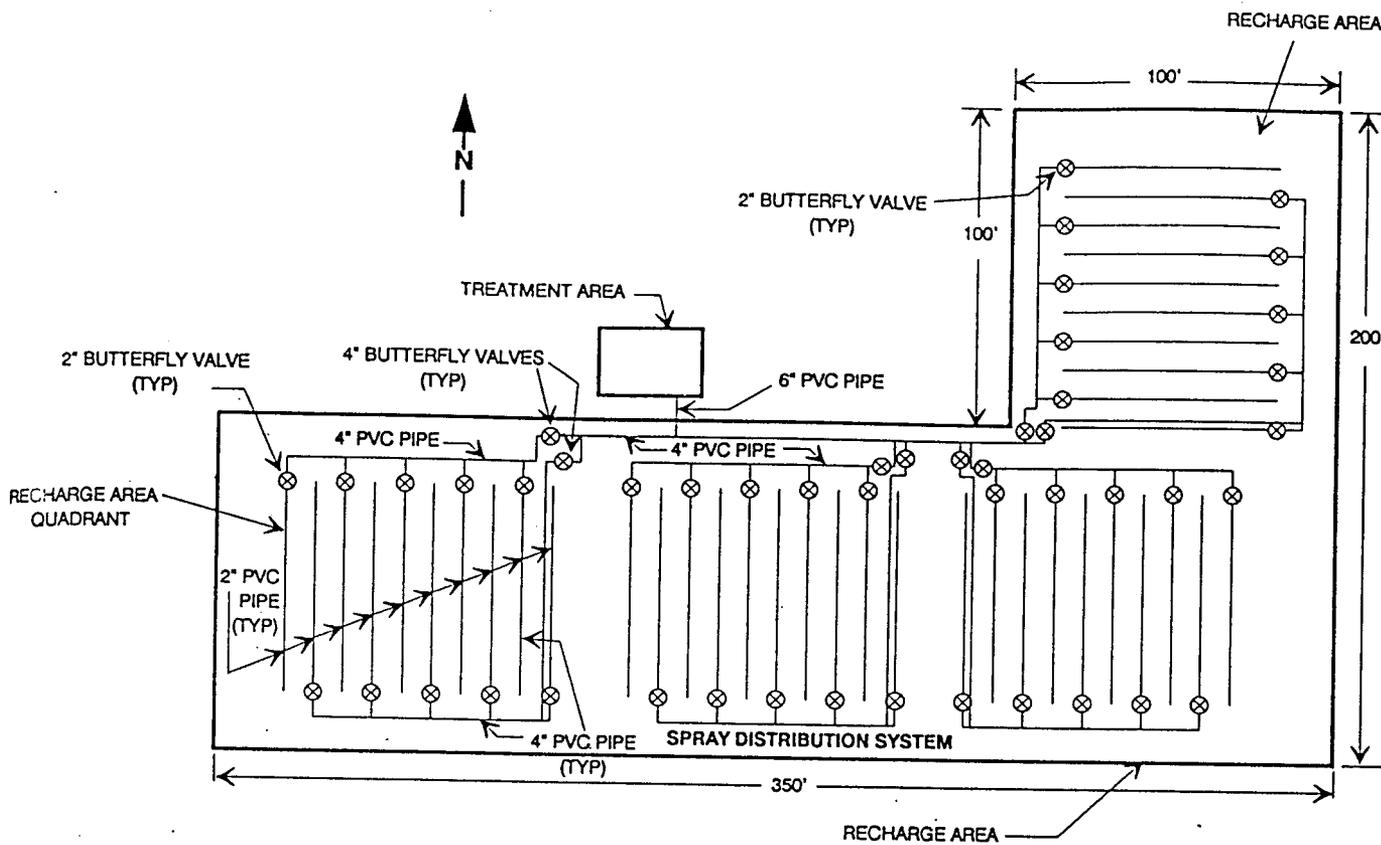


Figure 5. Recharge area layout.

flexible hoses, water in the interceptor basins is pumped until the basins are empty. Emptying the interceptor basins each time samples are collected ensures that the sampled water is representative of the treated water entering the soil at the time the samples are taken. A generalized cross section of the recharge area is presented in Figure 6.

Evaluation of System Performance

The remedial action system was designed and implemented in the summer of 1986, and startup testing of the system began in September 1986. After the winter shut-down and operating permit acquisition during spring, the system began to operate in the summer of 1987. The monitoring data obtained from July to October 1987 were used to evaluate system performance. These data were collected to meet regulatory monitoring requirements.

Monitoring requirements consisted of determining influent and effluent TCE concentrations and flow rates. The influent TCE concentration and flow rate were measured at the header pipe influent to the stripping tower. The header pipe carries combined flow from the 14 ground water purge wells. The effluent flow rate was measured at the pipe influent to the distribution system, and the TCE concentration was measured at the gravel bed-soil interface. The influent and effluent TCE concentration and the overall removal efficiency of the system are shown in Figure 7.

From the initial operating data, shown in Figure 7, the overall TCE removal efficiency of the system varied from 80 to 99 percent. The average TCE removal was 94.5

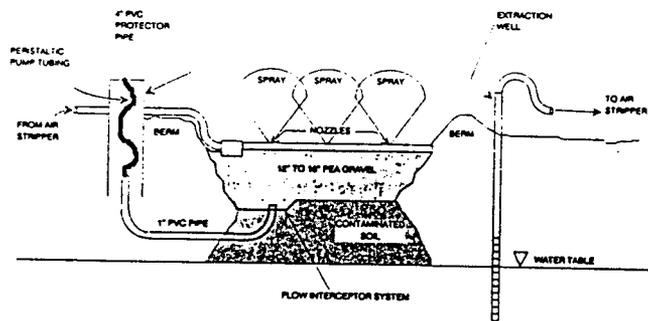


Figure 6. Generalized cross section recharge area.

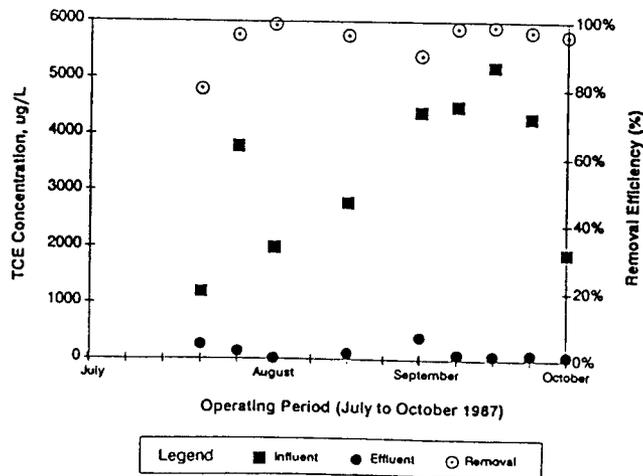


Figure 7. Influent and effluent TCE concentration data and overall system removal efficiency.

percent. This is slightly lower but in general agreement with the 97.3 to 99.1 percent removal efficiency observed during the pilot treatability study (Thomsen, K.O. et al. 1986). The data showed that approximately 75 percent TCE removal may be attributed to air stripping and an additional 20 percent to the spray recharge system.

From July to October 1987, approximately 24.5 million gallons of ground water were treated. Approximately 775 pounds of TCE were removed based on an average TCE concentration of 4000 $\mu\text{g/L}$ in the ground water entering the treatment system. If the system has been in operation for a full annual operating period (April through October), about 35 million gallons of ground water would have been treated, removing an estimated 1100 pounds of TCE.

It is apparent from the design of the system that flushing of the soil and containment of the ground water contaminant plume does occur. Unfortunately, the efficiency of the soil flushing and plume containment were not established because this was not required by the consent judgment.

Remedial Action Costs

The total capital cost of the remedial action was approximately \$3,000,000. This amount includes costs for site work, purge wells, stripping tower, piping, electrical controls and instrumentation, pumps, spray recharge, and gravel bed. The capital cost also includes fees for legal, permitting, engineering, and construction oversight services. The estimated annual O & M cost is \$145,000. This includes costs for labor, electrical power, maintenance, replacement parts, influent and effluent monitoring, permitting, and administration. Breakdown of the total capital cost and the estimated annual O & M costs is presented in Table 2.

The ground water will continue to be treated until the cleanup goal is met. The total period of operation could not be estimated because the efficiency of soil flushing and the effect on ground water conditions were not determined. Because the time required to remedy site contamination may be several years, the total annualized capital and O & M costs were estimated for three operating periods (15, 20, and 30 years) at a 10 percent discount rate. The annualized capital and O & M costs of the remedial action for treating 1000 gallons of ground water are \$1.70 for 15 years, \$1.60 for 20 years, and \$1.50 for 30 years.

Summary

To remedy contamination at a spent solvent disposal site, a ground water extraction, treatment, and recharge system was designed and implemented. The objectives of the remedial action were to: (1) capture the ground water contaminant plume to minimize off-site contaminant migration, (2) treat ground water until the remedial action goal (15 $\mu\text{g/L}$ of TCE in ground water) is met, and (3) flush contaminated soil with treated ground water to control the source of contamination and enhance site cleanup.

TABLE 2
Breakdown of Total Capital and Estimated Annual O & M Costs of Remediation System

	<u>Total Cost</u>
Capital Cost	
Site Work	\$ 300,000
Site cleanup, removal of debris, miscellaneous material handling, fencing, surveying, and access road construction	
Collection System	650,000
14 purge wells drilling and installation, pumps, instrumentation, piping and hydrogeologic studies	
Treatment System	500,000
Stripping tower, equipment delivery and installation, foundation work, discharge pumps, electrical controls and instrumentation, monitoring equipment, backwash tank, and accessories	
Recharge System	90,000
Building	80,000
Sampling Collection, Monitoring, and Laboratory Equipment	80,000
Miscellaneous Costs	<u>60,000</u>
Subtotal	\$ 1,760,000
Engineering (20 percent)	350,000
Construction Management, including health and safety during construction (15 percent)	270,000
Permitting and Legal	220,000
Contingencies (20 percent)	<u>400,000</u>
Total Capital Cost	\$3,000,000
Annual O & M Costs	
Pumps and Blowers (electrical)	\$ 10,000
System Maintenance and Material	25,000
Material	35,000
Monitoring	20,000
Permit Renewal	15,000
Administration	<u>20,000</u>
Subtotal	\$ 120,000
Contingencies	<u>25,000</u>
Total Annual O & M Costs	<u>\$ 145,000</u>

The initial operating data demonstrated that the system can effectively remove TCE from ground water (approximately 95 percent overall removal efficiency).

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The estimated annualized capital and O & M costs for treating 1000 gallons range from \$1.70 for 15 years to \$1.50 for 30 years.

References

Thomsen, K.O., B.H. Khara, and A.A. Aguwa. 1986. A collection/treatment/recharge/flushing ground water remediation program. In *Proceedings of the 7th National Conference on Management of Uncontrolled Hazardous Waste Sites*, Superfund '86, pp. 220-223.

Biographical Sketches

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Transport of organic contaminants in groundwater

Distribution and fate of chemicals in sand and gravel aquifers

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Frequent discoveries of groundwater contamination arising from use, storage, and disposal of hazardous chemicals underscore the need for an understanding of and ultimately the ability to predict the mechanisms of transport of contaminants in the subsurface. In particular, there has been a tremendous interest in developing computerized contaminant transport models, and many such models have been documented (1).

The ability to develop mathematical formulations of a problem that is not directly observable and the ease with which models can be used to predict contaminant migration have led to their widespread acceptance as important tools in the investigation of groundwater contamination in sand and gravel aquifers. However, it is critical to keep in mind that the strength of available models is directly related to the depth of present understanding of the fundamental processes that control the transport and fate of contaminants.

In this article we review the state of understanding of the physical, chemical, and biological processes that are thought to affect organic contaminants in the saturated (groundwater) zone. Although contaminants that enter the saturated zone often must first pass through the unsaturated (vadose) zone, we do not discuss that zone in any de-

tail. Because many aquifers used for water supply comprise horizontal geological units of sand and gravel, we confine our discussion to horizontal flow in such aquifers. Our purpose is to draw on the current understanding of the processes that are important to the saturated zone. This will help to illustrate the conceptual complexities of organic contaminant transport and distribution in the groundwater zone. Finally, we present some of the implications of groundwater contamination.

Transport processes

Organic contaminants can reach the groundwater zone either dissolved in water or as organic liquid phases that may be immiscible in water. Dissolved contaminants can result from spills or leaks of aqueous solutions or from the leaching of solid phases or immiscible organic liquids present in the vadose zone or land disposal areas. Organic liquids can be introduced to the subsurface by spills, leaks, or intentional disposal.

The subsurface transport of immiscible organic liquids is governed by a set of factors different from those for dissolved contaminants. However, some components of organic liquids can dissolve into the groundwater. Therefore, in the following synoptic discussion, we begin with the processes that most strongly influence the migration of dissolved organic solutes.

Advection

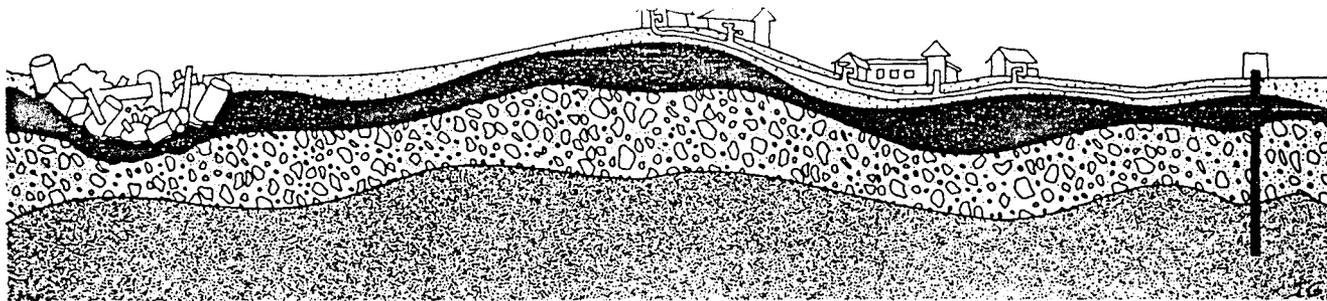
In sand and gravel aquifers, the dominant factor in the migration of a dissolved contaminant is advection, the process by which solutes are transported by the bulk motion of flowing groundwater. Groundwater generally flows from regions of the subsurface where water level is high to regions

where water level is low. Hydraulic gradient is the term used to describe the magnitude of this driving force. The average linear velocity at which groundwater flows through a granular medium, such as a sand and gravel aquifer, is equal to the product of the gradient and the inherent capability of the medium to transmit water. The latter is expressed as the ratio of the medium's hydraulic conductivity and porosity (2).

For uniform sand and gravel aquifers in gentle topography, the gradient normally follows the topography, and the groundwater flow rate can be estimated to within a factor of 10, given a general description of the geologic medium. More accurate estimates are possible if site-specific measurements of aquifer properties are available. Groundwater velocities in such aquifers typically range between 1 meter/year (m/y) and 1000 m/y. In most cases, however, the flow velocities under natural gradient conditions are probably between 10 m/y and 100 m/y. Thus, when monitoring wells or small supply wells in sand and gravel aquifers are located hundreds or thousands of meters downgradient of a contaminant source, the average travel time for the groundwater to flow from source to well typically is on the order of decades. In the zone of influence of a high-capacity well or wellfield, however, the artificially increased gradient substantially increases the local velocity, and the average travel times for groundwater flow are reduced.

Dispersion

Dissolved contaminants spread as they move with the groundwater. This process, called dispersion, results from two basic processes, molecular diffusion and mechanical mixing. The kinetic activity of dissolved solutes results in the net flux, or diffusion, of



the solutes from a zone of high concentration to a zone of lower concentration. Mechanical mixing results from variations in groundwater velocity within the porous aquifer caused by frictional forces, variations in pore geometry, and fluctuations in the local flow directions relative to the mean flow direction.

Dispersion and spreading during transport result in the dilution of contaminant pulses and the attenuation of concentration peaks; the maximum concentrations diminish with increasing distance from the source. In plumes containing irregular concentration distributions, this spreading will lead to an increase in plume uniformity with distance. In addition, dispersive spreading may result in the arrival of detectable contaminant concentrations at a given location significantly before the arrival time that is expected solely on the basis of the average groundwater flow rate.

Despite the practical importance of the dispersion process, there is currently no method to confidently predict the magnitude of dispersion for a previously unstudied field situation. For simple hydrogeological systems, the spreading is believed to be proportional to the flow rate. For more complex systems, the constant of proportionality—termed the dispersivity—appears to depend on the structure of the geologic medium in such a manner that it varies with the distance traversed (3, 4). Furthermore, dispersion in the direction of flow often is observed to be markedly greater than dispersion in the directions transverse to the flow. Although very little is known about vertical transverse dispersivity, past field work on aquifers of sedimentary origin indicates that the ratio of apparent dispersivities in the longitudinal and horizontal transverse directions typically ranges from 10 to 100 (3, 4).

In the absence of detailed studies to determine the dispersive characteristics of a given field situation, longitudinal and transverse dispersivities must be estimated based on prior field work in similar hydrogeological systems. Given the current level of understanding, such estimates are characterized by a high degree of uncertainty. For example, longitudinal dispersivity values inferred

from past field efforts range over approximately one order of magnitude when they are compared for a given transport distance (3). Furthermore, for many problems pertaining to plume development and extent, continuing research suggests that dispersion in the transverse directions—about which we know even less—is of more practical importance.

Sorption and retardation

Some dissolved contaminants may interact with the aquifer solids encountered along the flow path through adsorption, partitioning, ion exchange, and other processes (2). These interactions result in the contaminants' distribution between the aqueous phase and the aquifer solids, diminution of concentrations in the aqueous phase, and retardation of the movement of the contaminant relative to groundwater flow (5-8). The higher the fraction of the contaminant sorbed, the more retarded is its transport.

For some contaminants, such as ionic species of heavy metals (cadmium, chromium, etc.) and certain organic solutes, the degree of interaction depends on many factors. These include the concentration and characteristics of the contaminant, the characteristics of the aquifer solids, the pH of the groundwater, and the presence of other dissolved constituents (9-11). It is possible for the degree of interaction, and therefore retardation, to vary in space and time due to variations in one or more of these factors in the natural groundwater environment (10, 11). Complexities such as these confound the prediction of groundwater transport of such contaminants.

In the case of some groundwater contaminants, such as certain halogenated organic solvents, the interaction, called sorption, is often affected in a significant way by only two factors: the contaminant's hydrophobicity (its antipathy to dissolving in water) and the fraction of solid organic matter in the aquifer solids (organic carbon content) (9). In a homogeneous aquifer, therefore, sorption of a hydrophobic organic solute should theoretically be constant in space and time. If the sorptive interaction is at equilibrium and completely

reversible, the solute should move at a constant average velocity equal to the groundwater's average velocity, divided by the "retardation factor." Such a contaminant is said to be linearly retarded.

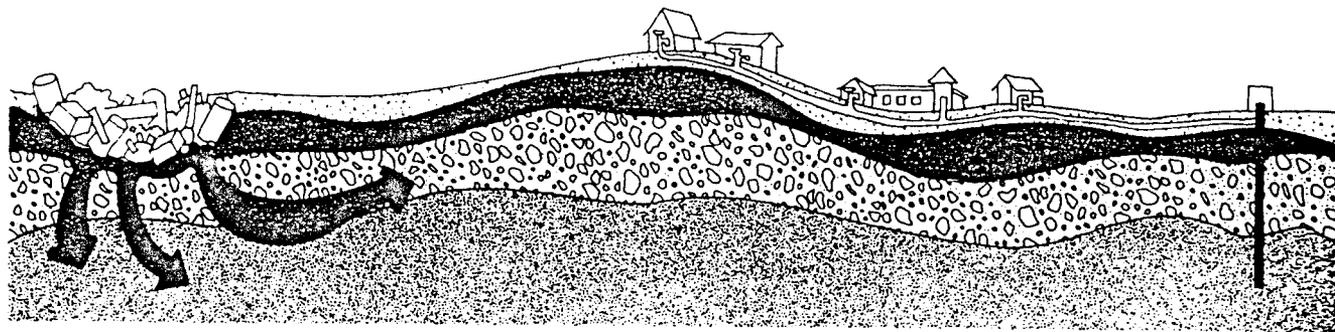
More hydrophobic compounds should be more highly retarded, a trend observed in several field studies (12-14). Roberts et al. show that retardation factors for hydrophobic organic contaminants of concern can be expected to range over four orders of magnitude, from slightly greater than one to as much as 10,000 (15). Among the most common groundwater contaminants are carbon tetrachloride, 1,1,1-trichloroethane (TCA), trichloroethylene (TCE), and tetrachloroethylene (perchloroethylene, PCE) (16). The retardation factors for these compounds are expected to fall in a range from 1 to 10 for many sand and gravel aquifers that are low in solid organic matter. Thus, in many aquifers used for water supply these contaminants would be expected to migrate at rates from 10% to nearly 100% of the velocity of the groundwater itself.

As mentioned above, however, the simple hydrophobic sorption and retardation model may not be applicable in all cases. For example, sorption by mineral surfaces may approach or exceed that by the solid organic matter if the ratio of mineral surface area to organic matter content is large (9, 17). In addition, there is evidence that in some cases sorption equilibrium may require weeks or months and thus may not always be reached in the field (9).

Variability of aquifer properties and interaction between sorption and other processes also may confound the application of the simple retardation concepts. For example, in heterogeneous sand and gravel aquifers the highest hydraulic conductivity zones may have the least solid organic matter. In such a case, the lowest retardation of hydrophobic organic contaminants would occur in the zones of fastest groundwater flow. Predictions of contaminant migration that do not account for this hydrogeologic heterogeneity can be significantly in error.

Chemical, biological transformation

Organic contaminants can be transformed into other compounds by an ex-



traordinarily complex set of chemical and biological mechanisms. The effects, relative importance, and interactions of these processes in the groundwater zone, which are not well understood, are increasingly the subject of research.

The principal classes of chemical reactions that can affect organic contaminants in water are hydrolysis and oxidation (18). Although empirical methods have been developed to estimate the rate constants for the effects of both processes on particular contaminants under specific solution conditions, the applicability of these methods to reactions in the groundwater zone is unknown. It is believed, however, that most chemical reactions occurring in the groundwater zone are likely to be slow compared with transformations mediated by microorganisms (19, 20).

There is good evidence that certain organic groundwater contaminants can be biologically transformed by microorganisms attached to solid surfaces within the aquifer (19). The attached bacteria obtain energy and nutrients from the groundwater flowing by and may form biofilms as their numbers increase. Energy for growth is obtained from oxidation of organic substrates or inorganic compounds, such as hydrogen or reduced forms of iron, nitrogen, or sulfur (19). Microorganisms vary in their ability to use the different electron acceptors required for these oxidations: Some use oxygen available under aerobic groundwater conditions. Others may use nitrate, sulfate, or carbon dioxide when conditions are anoxic (19).

There are many factors affecting the rates of biotransformation of organic compounds, including water temperature and pH, the number and species of microorganisms present, the concentration of the substrate, the presence of microbial toxicants and nutrients, and the availability of electron acceptors (21). In some cases, the native microflora may not be able to transform a specific compound, or may manifest that ability only after a considerable period of acclimation. It is not yet known whether deep aquifers generally contain sufficient numbers of bacteria to achieve substantial biotransformation rates of organic contaminants, but

Wilson and McNabb have found surprisingly high numbers of bacteria in shallow, unconfined aquifers at depths of 6 m or less (22).

There is a minimum concentration to which a single substrate can be decomposed under steady-state conditions. Below this level there is insufficient energy available to support continued bacterial growth (21). Biodegradable organic contaminants often are present at trace concentrations below the minimum level. In such cases biotransformation of the contaminants can occur if they are used as secondary substrates, but this requires the presence of an abundant primary substrate (or combination of degradable primary substrates) and bacteria that are able to transform both primary and secondary substrates (21).

It is widely believed that biotransformation of trace organic contaminants can and does occur in the groundwater zone under some conditions, sometimes after acclimation periods of months or years. The rates are believed to range widely, with half-lives ranging from a few days to many years, and may be significant in light of the low groundwater flow rates and long residence times that characterize aquifers (21).

Nonetheless, transformation of a toxic organic solute is no assurance that it has been converted to harmless or even less hazardous products. Biotransformation of common groundwater contaminants, such as PCE, TCE, and TCA, can result in the formation of such intermediates as vinyl chloride, which cannot be further transformed under prevailing conditions (21). Given our limited understanding of transformation processes and the factors influencing them, prudence dictates that in forecasting the effects of groundwater contamination, hazardous contaminants must be assumed, in the absence of site-specific evidence to the contrary, to persist indefinitely.

Immiscible organic liquids

Organic compounds differ widely in their solubility, from infinitely miscible polar compounds, such as methanol, to extremely low solubility nonpolar compounds, such as polynuclear aromatic

hydrocarbons (23). Commonly encountered groundwater contaminants, including halogenated aliphatics with one or two carbon atoms, tend to have moderately low solubilities (<1%). Thus many organic liquids released to the subsurface may migrate as discrete nonaqueous phases, some components of which may dissolve into the surrounding groundwater. The migration of an immiscible organic liquid phase in the subsurface is governed largely by its density and viscosity.

Density differences of about 1% are known to influence fluid movement significantly in the subsurface environment. For example, the stratification of saltwater and freshwater occurs at a density difference of 3.5%. With few exceptions, the densities of organic liquids differ from that of water by more than 1%. In most cases the difference is more than 10%. The specific gravities of hydrocarbons (gasoline and other petroleum distillates) may be as low as 0.7, and halogenated hydrocarbons are almost without exception significantly more dense than water. Chlorinated aliphatic compounds containing one- and two-carbon atoms have specific gravities from 1.2 to 1.5.

It is convenient to consider organic liquids less dense than water as "floaters," which spread across the water table, and organic liquids more dense than water as "sinkers," which may plummet through sand and gravel aquifers to the underlying aquitard (relatively impermeable layers) where present. There is extensive evidence from field studies that low-density organic liquids float on the water table (24). The sinking phenomenon has been demonstrated in physical model experiments by Schwille (25), and some corroborative evidence has been found in field observations of the spatial distribution of contaminants near landfills and other sources (26). It is important to recognize that the migration of dense organic liquids is largely uncoupled from the hydraulic gradient that drives advective transport and that the movement may have a dominant vertical component even in horizontally flowing aquifers.

The transport of an organic liquid phase also is influenced by its viscosity

and its surface-wetting properties compared with those of water. Schwille has shown that halogenated aliphatics tend to spread by capillary action into aquifer media and that they tend to be retained in amounts of about 0.3% to 5% by volume, following the passage of the organic liquid (25). This points to the possibility of storage of large quantities of immiscible liquid organic contaminants as droplets dispersed within the pores of aquifer media, even if the bulk of the migrating mass of liquid is removed. The organic liquid droplets retained in the aquifer may then dissolve over time into the groundwater flowing past them.

An organic liquid of moderately low solubility (such as PCE) can contaminate as much as 10,000 times its own volume to its solubility limit. However, organic compounds are only rarely found in groundwater at concentrations approaching their solubility limits, even when organic liquid phases are known or suspected to be present. The observed concentrations are usually more than a factor of 10 lower than the solubility, presumably because of the diffusional limitations of dissolution and the dilution of the dissolved organic contaminants by dispersion. This implies that the volume of groundwater that could be contaminated by an organic liquid phase is much larger than that calculated by assuming dissolution to the solubility limit. It is evident that what might once have been considered a small spill or leak (for example, tens of gallons of a pure industrial solvent spilled every time a tank is filled or a transfer line is flushed) may in fact constitute a significant source of contamination if the spilled liquid reaches the groundwater zone.

Transport and distribution

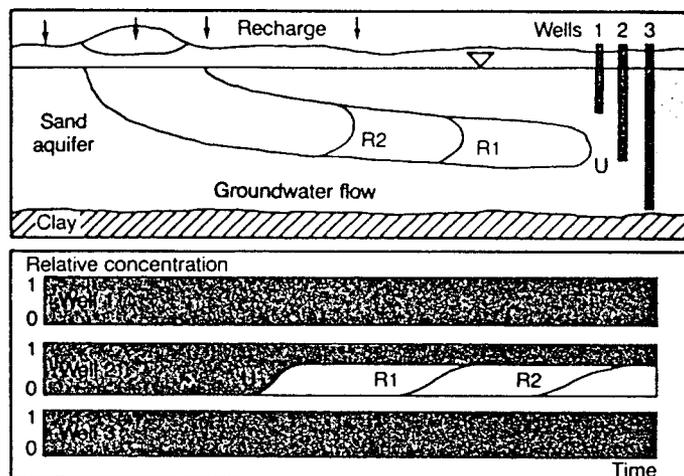
After discussing the processes that affect organic contaminant transport in the subsurface, it is worthwhile to consider illustrations of their effects in the conceptually simplest hydrogeologic domain: a uniform, unconfined sandy aquifer underlain by a level horizontal aquitard. We will assume that the water table is close to the surface and that the hazardous organic chemicals have been released directly into the sand by a spill or leak or by the leaching of materials deposited in an unlined dump.

Continuous sources

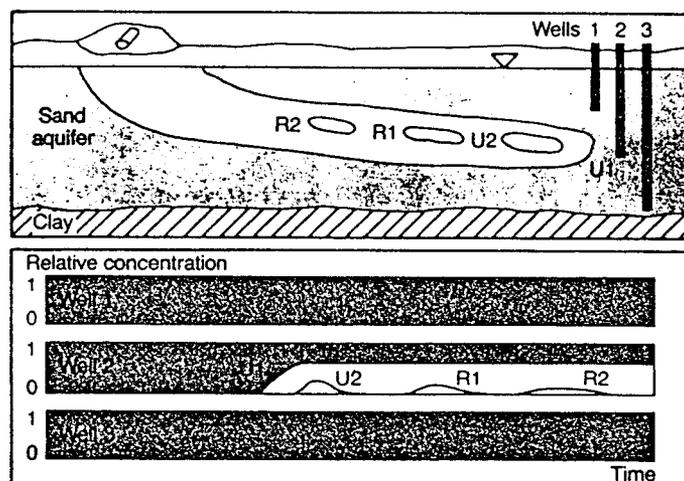
Figure 1a illustrates a case in which hazardous chemicals are distributed uniformly in a waste mass or contaminated soil zone and are leached slowly by precipitation. We assume that the aqueous leachate is relatively uniform in composition and flow over many years (implying a large reservoir of

FIGURE 1
Contamination from various sources in an unconfined aquifer*

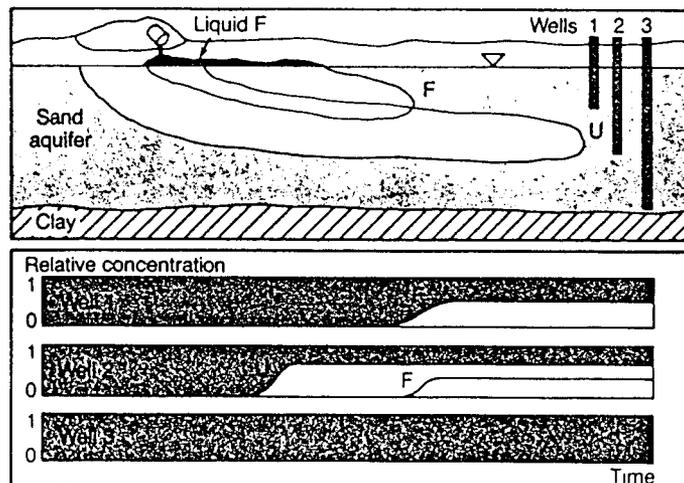
(a) Continuous source of three dissolved contaminants; unretarded (U) and retarded to varying degrees (R1, R2)



(b) Continuous source of a dissolved, unretarded contaminant (U1) and a pulse source of three dissolved contaminants; unretarded (U2) and retarded (R1, R2)



(c) Continuous source of a dissolved, unretarded contaminant (U) and a pulse source of an organic liquid (F) that floats on and slowly dissolves into the groundwater



*Top frames show contaminant distribution at one point in time. Bottom frames show concentration history at each well. Relative concentration expresses the observed value as a fraction of the undiluted leachate (U1, U2, R1, R2), contaminant solubility (F, S), or parent compound concentration (SS).

leachable contaminants). We also assume that the leachate contains non-sorbing, unretarded contaminants (U), such as chloride ions, and two contaminants that are linearly retarded to varying degrees (R1 and R2). Examples might be TCE and PCE.

Figure 1a shows the expected contaminant distributions at one point in time and the contaminant concentration histories observed in three monitoring wells screened at different depths. Intermittent pumping of the wells is assumed to withdraw minor amounts of water and does not influence plume migration. The vertical gradient, owing to natural recharge or density effects, results in the contaminants being observed in Well 2 only. It is also apparent that the contaminants arrive at the well

at different times because the plume from the source is not spatially uniform, even in this simple case, but actually is composed of three overlapping plumes migrating at different rates.

The concentrations of all contaminants observed in Well 2 rise to levels below those of the undiluted leachate. The gradual rise is the result of dispersion (primarily longitudinal) of the advancing front. The reduction in maximum concentration below the undiluted value is because of dispersion in the transverse directions.

Pulse sources

A different case is illustrated in Figure 1b. Here we assume that a mass of uniform bulk waste produces aqueous leachate that is uniform in flow and

composition but contains only nonsorbing contaminants (U1). This bulk waste contains a local source that releases a slug containing three contaminants (one is conservative and two are linearly retarded—U2, R1, and R2, respectively) shortly after leaching of the bulk waste begins. The release could be the result of the sudden rupture of a buried drum (due to corrosion or compaction), leaching of a localized contaminant source over a relatively short time (from a load of contaminated soil), or disposal of a tank load of an aqueous solution of contaminants.

Figure 1b shows contaminant pulses migrating within the larger plume emanating from the bulk waste. Note that the pulses lengthen with distance, because of dispersion. As before, only Well 2 is affected, but the concentration history at the well is different. Contaminants U2, R1, and R2 affect the monitoring well for finite periods that depend on the duration and size of the source and the advective-dispersive characteristics of the aquifer. Note also that dispersion of the pulse of conservative contaminant U2 will reduce its peak concentration observed at Well 2 to substantially below that of the undiluted initial slug. For the retarded contaminants, both dispersion and sorption serve to reduce the observed peak solution concentrations.

Low-density organic liquids

Suppose now that there is a similar situation, except that the local source within the bulk waste creates a release of an organic liquid contaminant, such as gasoline, that is immiscible with and less dense than the groundwater. In this case, the organic liquid migrates vertically to and then floats on the water table, spreading out in the downgradient direction. If the organic liquid contains contaminant F (e.g., benzene), which is slightly soluble in water, a plume will develop in the saturated zone.

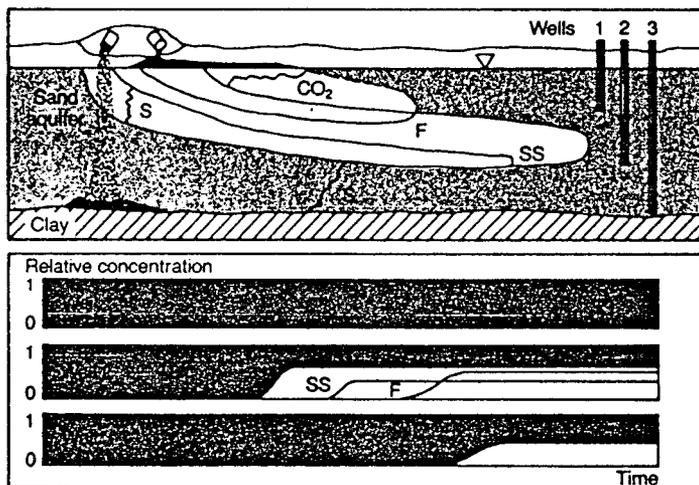
Figure 1c shows the overlapping plumes that might result. It also shows that the dissolving contaminant might affect both of the shallow wells, depending on the size of the source and the degree to which the organic liquid spreads as it floats on the water table. The dissolved contaminant F is assumed to be slightly retarded during transport in the saturated zone. The concentration of F observed in the wells is likely to be considerably lower than its solubility in water as a result of dispersion and sorption.

High-density organic liquids

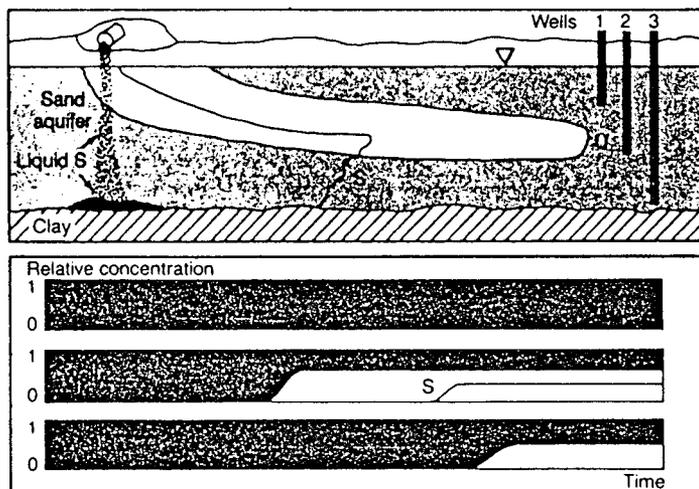
The next case is similar to that just described, except that a small source within the bulk waste releases an or-

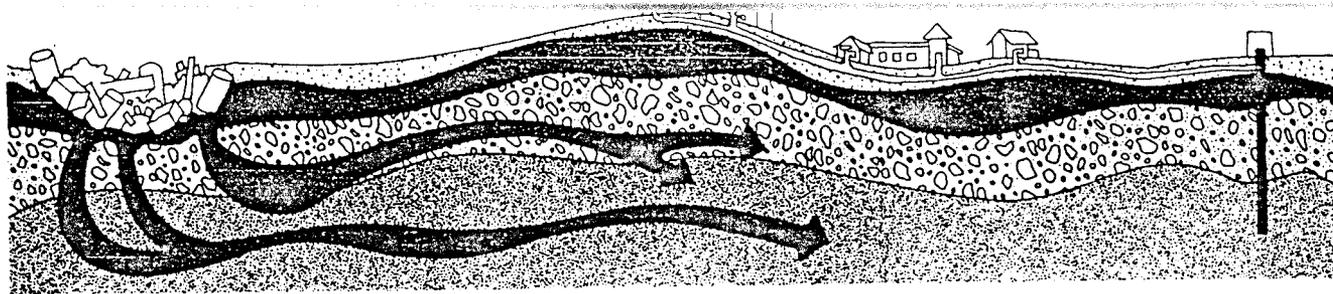
FIGURE 1 (continued)

(d) Continuous source of a dissolved, unretarded contaminant (U) and a pulse source of an organic liquid (S) that sinks through and slowly dissolves into the groundwater



(e) Continuous source of dissolved, unretarded solutes (U) and two pulse sources: floater (F) and sinker (S). The aquifer is aerobic except within the anaerobic U plume. Dissolved F degrades aerobically to CO₂ and water. Dissolved S degrades anaerobically to by-product SS, which is not readily degraded and is less retarded than S or F





ganic liquid contaminant S (e.g., TCE), which is immiscible with and more dense than the groundwater. Liquid S is assumed to sink through the saturated zone as an immiscible phase, displacing the groundwater as it descends, as shown in Figure 1d. Some of the organic liquid is retained by capillary forces in the pores of the aquifer material. If the contaminant is slightly soluble in water, a plume develops by dissolution of the contaminant liquid retained in the aquifer pores as well as by dissolution of the pool of contaminant liquid residing on the bottom of the aquifer.

The resulting plume shape may be complex, depending on the speed of the organic liquid's descent and the amount retained in the aquifer pores. Note that the clay aquitard is assumed level and impervious to the contaminant liquid. Thus, there is no migration of the contaminant pool along the aquifer other than spreading. Figure 1d shows that the two deeper wells will be affected by the contaminant. Also note that the observed concentrations will rise to some value well below the contaminant's solubility. In addition, the contaminant concentration may remain relatively constant for long periods, even when the volume spilled does not seem particularly large from a practical standpoint (a few hundred or thousand gallons of slightly soluble organic liquid contaminants).

Transformable compounds

In a somewhat more complicated case, consider the simple sand-gravel hydrogeologic setting affected by several organic contaminants with different potentials for biotransformation. Assume the leachate from the bulk waste contains readily biodegradable organic solutes. Assume also that the resultant microbiological growth in the aquifer renders the traversed zone anaerobic, as shown by the nonsorbing tracer U in Figure 1e. The remainder of the saturated zone stays aerobic. A small source releases a low-density organic liquid F, which is slightly soluble. Once contaminant F is dissolved, it can be completely mineralized (transformed to carbon dioxide) under aerobic conditions, but it is not biotrans-

formable under anaerobic conditions. Another small source releases an organic liquid "sinker" S, which is slightly soluble. The dissolved contaminant S is not transformable aerobically, but can be biotransformed under anaerobic conditions to an intermediate SS, which is more mobile than S and not readily degradable (for example, TCE transformed anaerobically to vinyl chloride).

In this case, contaminant F disappears from the aerobic zone above the leachate plume but persists within the plume. Conversely, contaminant S persists below the bulk leachate plume but disappears within the plume. Contaminant SS, which would not be found in the waste source itself, appears within the bulk waste leachate plume as a result of biotransformation. In this simplified portrayal no contaminants are observed in Well 1. Well 2 is affected by the bulk leachate plume U, the persistent portion of the F plume, and the transformation product SS. Well 3 is affected by contaminant S only.

Transport time scales

To illustrate the phenomena in a relatively clear and general fashion, the schematic diagrams in Figure 1 do not show specific time and distance scales. In fact, it is difficult to provide such scales with certainty because of the gaps in our understanding of the fundamental processes as they operate under natural conditions. However, if we take a situation that is sufficiently idealized to allow the application of a simple transport model, we can create and examine a quantitative illustration of the effects of advection, dispersion, and sorption.

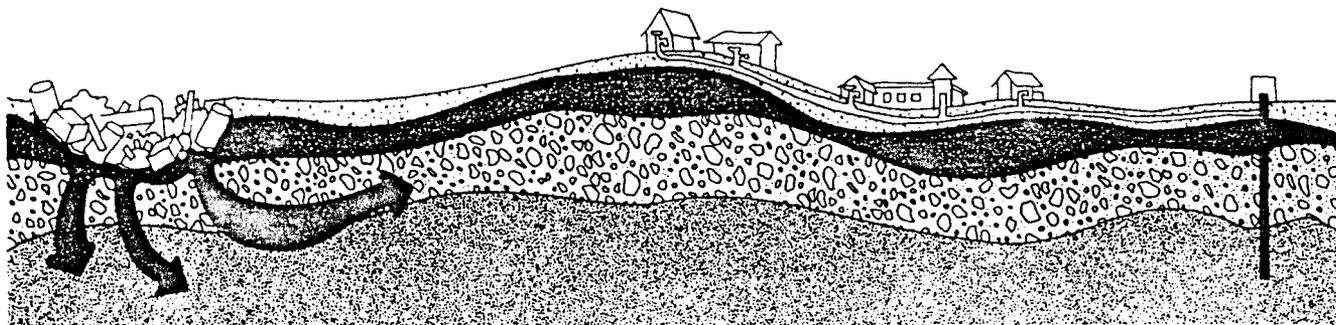
We assume a sandy aquifer in which the average groundwater flow rate is 45 m/y, within the velocity range that is common. The aquifer is horizontal, unconfined, and has a 10-m-thick saturated zone as indicated in Figure 2a. There is an unlined waste burial pit that is 50 m wide (crossgradient) and a solvent tank buried in a 5-m-wide unlined vault immediately adjacent to it. The bottom of each (the aquifer and the pit) is very near the water table. The waste pit contains chloride ions (Cl^-), TCE, and PCE, which leach continuously.

The leachate contaminates the entire 10-m depth of the aquifer below the impoundment and results in an essentially constant and vertically uniform concentration of the contaminants at $x = 0$. We will assume, for simplicity, that the concentration of each is 1 ppm (1000 ppb) at $x = 0$; this is less than 0.1% and 1% of the solubilities of TCE and PCE, respectively.

We also will assume that simultaneous with the onset of contamination from the waste pit, a small spill occurs as the solvent tank is being filled, and 15 gal (57 L) of pure TCA sinks rapidly through the bottom of the vault, uniformly contaminating the saturated zone in an area 5 m wide. The liquid droplets of TCA in the aquifer then dissolve into the groundwater flowing by, and the TCA concentration reaches 500 ppm, approximately one-half its solubility, before dispersion and dilution begin beyond $x = 0$. Given the volume spilled, the area contaminated by the organic liquid, the initial concentration, the groundwater flow rate, and other typical aquifer characteristics, we calculate that complete dissolution of the TCA droplets takes approximately 10 weeks. It is worth noting, however, that dissolution times on the order of years can be calculated for industrial spills that would still be considered small (hundreds or thousands of gallons).

The dissolved contaminants migrate toward a monitoring well directly downgradient at a distance of 1000 m. Pumping the well during sampling is assumed to have a negligible effect on the gradient. We assume moderate dispersive characteristics for such migration distances in sandy aquifers: 10 m and 1 m for longitudinal and transverse horizontal dispersivity, respectively (3). Also we assume that the chloride ion is not retarded and the retardation factors for TCE, TCA, and PCE are 2, 2, and 3, respectively. Such retardation factors would be expected for transport of these solutes in a sandy aquifer of about 0.1% carbon content (15).

In this highly idealized case, we would expect the water quality history at the monitoring well to be that shown in Figure 2b. The vertical dashed line at about 22 years denotes the solute arrival time expected solely on the basis



traordinarily complex set of chemical and biological mechanisms. The effects, relative importance, and interactions of these processes in the groundwater zone, which are not well understood, are increasingly the subject of research.

The principal classes of chemical reactions that can affect organic contaminants in water are hydrolysis and oxidation (18). Although empirical methods have been developed to estimate the rate constants for the effects of both processes on particular contaminants under specific solution conditions, the applicability of these methods to reactions in the groundwater zone is unknown. It is believed, however, that most chemical reactions occurring in the groundwater zone are likely to be slow compared with transformations mediated by microorganisms (19, 20).

There is good evidence that certain organic groundwater contaminants can be biologically transformed by microorganisms attached to solid surfaces within the aquifer (19). The attached bacteria obtain energy and nutrients from the groundwater flowing by and may form biofilms as their numbers increase. Energy for growth is obtained from oxidation of organic substrates or inorganic compounds, such as hydrogen or reduced forms of iron, nitrogen, or sulfur (19). Microorganisms vary in their ability to use the different electron acceptors required for these oxidations: Some use oxygen available under aerobic groundwater conditions. Others may use nitrate, sulfate, or carbon dioxide when conditions are anoxic (19).

There are many factors affecting the rates of biotransformation of organic compounds, including water temperature and pH, the number and species of microorganisms present, the concentration of the substrate, the presence of microbial toxicants and nutrients, and the availability of electron acceptors (21). In some cases, the native microflora may not be able to transform a specific compound, or may manifest that ability only after a considerable period of acclimation. It is not yet known whether deep aquifers generally contain sufficient numbers of bacteria to achieve substantial biotransformation rates of organic contaminants, but

Wilson and McNabb have found surprisingly high numbers of bacteria in shallow, unconfined aquifers at depths of 6 m or less (22).

There is a minimum concentration to which a single substrate can be decomposed under steady-state conditions. Below this level there is insufficient energy available to support continued bacterial growth (21). Biodegradable organic contaminants often are present at trace concentrations below the minimum level. In such cases biotransformation of the contaminants can occur if they are used as secondary substrates, but this requires the presence of an abundant primary substrate (or combination of degradable primary substrates) and bacteria that are able to transform both primary and secondary substrates (21).

It is widely believed that biotransformation of trace organic contaminants can and does occur in the groundwater zone under some conditions, sometimes after acclimation periods of months or years. The rates are believed to range widely, with half-lives ranging from a few days to many years, and may be significant in light of the low groundwater flow rates and long residence times that characterize aquifers (21).

Nonetheless, transformation of a toxic organic solute is no assurance that it has been converted to harmless or even less hazardous products. Biotransformation of common groundwater contaminants, such as PCE, TCE, and TCA, can result in the formation of such intermediates as vinyl chloride, which cannot be further transformed under prevailing conditions (21). Given our limited understanding of transformation processes and the factors influencing them, prudence dictates that in forecasting the effects of groundwater contamination, hazardous contaminants must be assumed, in the absence of site-specific evidence to the contrary, to persist indefinitely.

Immiscible organic liquids

Organic compounds differ widely in their solubility, from infinitely miscible polar compounds, such as methanol, to extremely low solubility nonpolar compounds, such as polynuclear aromatic

hydrocarbons (23). Commonly encountered groundwater contaminants, including halogenated aliphatics with one or two carbon atoms, tend to have moderately low solubilities (<1%). Thus many organic liquids released to the subsurface may migrate as discrete nonaqueous phases, some components of which may dissolve into the surrounding groundwater. The migration of an immiscible organic liquid phase in the subsurface is governed largely by its density and viscosity.

Density differences of about 1% are known to influence fluid movement significantly in the subsurface environment. For example, the stratification of saltwater and freshwater occurs at a density difference of 3.5%. With few exceptions, the densities of organic liquids differ from that of water by more than 1%. In most cases the difference is more than 10%. The specific gravities of hydrocarbons (gasoline and other petroleum distillates) may be as low as 0.7, and halogenated hydrocarbons are almost without exception significantly more dense than water. Chlorinated aliphatic compounds containing one- and two-carbon atoms have specific gravities from 1.2 to 1.5.

It is convenient to consider organic liquids less dense than water as "floaters," which spread across the water table, and organic liquids more dense than water as "sinkers," which may plummet through sand and gravel aquifers to the underlying aquitard (relatively impermeable layers) where present. There is extensive evidence from field studies that low-density organic liquids float on the water table (24). The sinking phenomenon has been demonstrated in physical model experiments by Schuille (25), and some corroborative evidence has been found in field observations of the spatial distribution of contaminants near landfills and other sources (26). It is important to recognize that the migration of dense organic liquids is largely uncoupled from the hydraulic gradient that drives advective transport and that the movement may have a dominant vertical component even in horizontally flowing aquifers.

The transport of an organic liquid phase also is influenced by its viscosity

and its surface-wetting properties compared with those of water. Schwille has shown that halogenated aliphatics tend to spread by capillary action into aquifer media and that they tend to be retained in amounts of about 0.3% to 5% by volume, following the passage of the organic liquid (25). This points to the possibility of storage of large quantities of immiscible liquid organic contaminants as droplets dispersed within the pores of aquifer media, even if the bulk of the migrating mass of liquid is removed. The organic liquid droplets retained in the aquifer may then dissolve over time into the groundwater flowing past them.

An organic liquid of moderately low solubility (such as PCE) can contaminate as much as 10,000 times its own volume to its solubility limit. However, organic compounds are only rarely found in groundwater at concentrations approaching their solubility limits, even when organic liquid phases are known or suspected to be present. The observed concentrations are usually more than a factor of 10 lower than the solubility, presumably because of the diffusional limitations of dissolution and the dilution of the dissolved organic contaminants by dispersion. This implies that the volume of groundwater that could be contaminated by an organic liquid phase is much larger than that calculated by assuming dissolution to the solubility limit. It is evident that what might once have been considered a small spill or leak (for example, tens of gallons of a pure industrial solvent spilled every time a tank is filled or a transfer line is flushed) may in fact constitute a significant source of contamination if the spilled liquid reaches the groundwater zone.

Transport and distribution

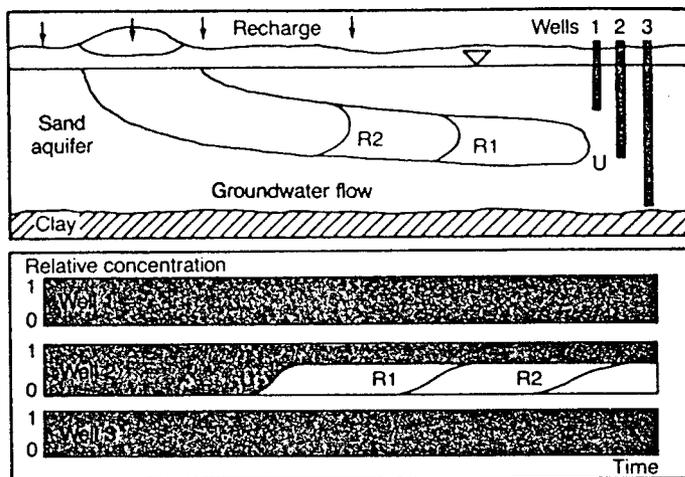
After discussing the processes that affect organic contaminant transport in the subsurface, it is worthwhile to consider illustrations of their effects in the conceptually simplest hydrogeologic domain: a uniform, unconfined sandy aquifer underlain by a level horizontal aquitard. We will assume that the water table is close to the surface and that the hazardous organic chemicals have been released directly into the sand by a spill or leak or by the leaching of materials deposited in an unlined dump.

Continuous sources

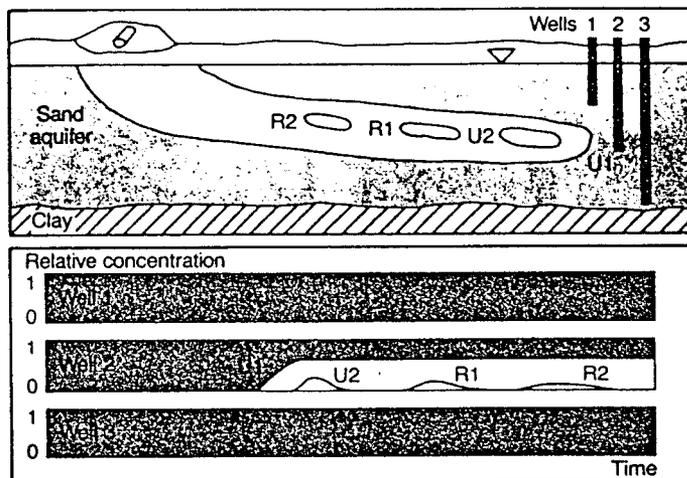
Figure 1a illustrates a case in which hazardous chemicals are distributed uniformly in a waste mass or contaminated soil zone and are leached slowly by precipitation. We assume that the aqueous leachate is relatively uniform in composition and flow over many years (implying a large reservoir of

FIGURE 1 Contamination from various sources in an unconfined aquifer*

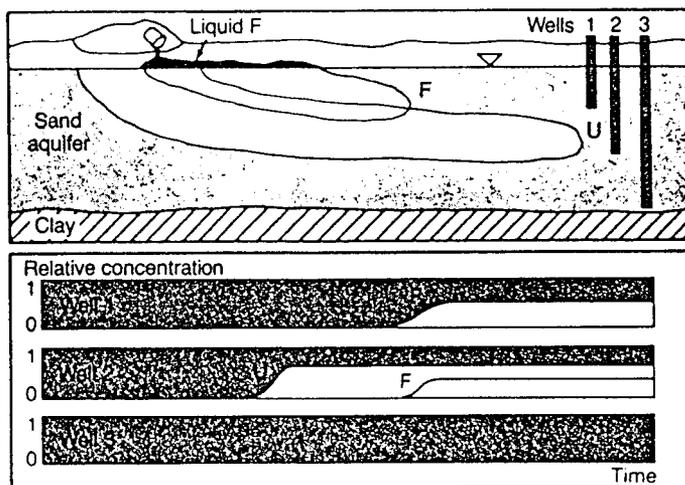
(a) Continuous source of three dissolved contaminants; unretarded (U) and retarded to varying degrees (R1, R2)



(b) Continuous source of a dissolved, unretarded contaminant (U1) and a pulse source of three dissolved contaminants; unretarded (U2) and retarded (R1, R2)



(c) Continuous source of a dissolved, unretarded contaminant (U) and a pulse source of an organic liquid (F) that floats on and slowly dissolves into the groundwater



*Top frames show contaminant distribution at one point in time. Bottom frames show concentration history at each well. Relative concentration expresses the observed value as a fraction of the undiluted leachate (U1, U2, R1, R2), contaminant solubility (F, S), or parent compound concentration (SS).

leachable contaminants). We also assume that the leachate contains non-sorbing, unretarded contaminants (U), such as chloride ions, and two contaminants that are linearly retarded to varying degrees (R1 and R2). Examples might be TCE and PCE.

Figure 1a shows the expected contaminant distributions at one point in time and the contaminant concentration histories observed in three monitoring wells screened at different depths. Intermittent pumping of the wells is assumed to withdraw minor amounts of water and does not influence plume migration. The vertical gradient, owing to natural recharge or density effects, results in the contaminants being observed in Well 2 only. It is also apparent that the contaminants arrive at the well

at different times because the plume from the source is not spatially uniform, even in this simple case, but actually is composed of three overlapping plumes migrating at different rates.

The concentrations of all contaminants observed in Well 2 rise to levels below those of the undiluted leachate. The gradual rise is the result of dispersion (primarily longitudinal) of the advancing front. The reduction in maximum concentration below the undiluted value is because of dispersion in the transverse directions.

Pulse sources

A different case is illustrated in Figure 1b. Here we assume that a mass of uniform bulk waste produces aqueous leachate that is uniform in flow and

composition but contains only non-sorbing contaminants (U1). This bulk waste contains a local source that releases a slug containing three contaminants (one is conservative and two are linearly retarded—U2, R1, and R2, respectively) shortly after leaching of the bulk waste begins. The release could be the result of the sudden rupture of a buried drum (due to corrosion or compaction), leaching of a localized contaminant source over a relatively short time (from a load of contaminated soil), or disposal of a tank load of an aqueous solution of contaminants.

Figure 1b shows contaminant pulses migrating within the larger plume emanating from the bulk waste. Note that the pulses lengthen with distance, because of dispersion. As before, only Well 2 is affected, but the concentration history at the well is different. Contaminants U2, R1, and R2 affect the monitoring well for finite periods that depend on the duration and size of the source and the advective-dispersive characteristics of the aquifer. Note also that dispersion of the pulse of conservative contaminant U2 will reduce its peak concentration observed at Well 2 to substantially below that of the undiluted initial slug. For the retarded contaminants, both dispersion and sorption serve to reduce the observed peak solution concentrations.

Low-density organic liquids

Suppose now that there is a similar situation, except that the local source within the bulk waste creates a release of an organic liquid contaminant, such as gasoline, that is immiscible with and less dense than the groundwater. In this case, the organic liquid migrates vertically to and then floats on the water table, spreading out in the downgradient direction. If the organic liquid contains contaminant F (e.g., benzene), which is slightly soluble in water, a plume will develop in the saturated zone.

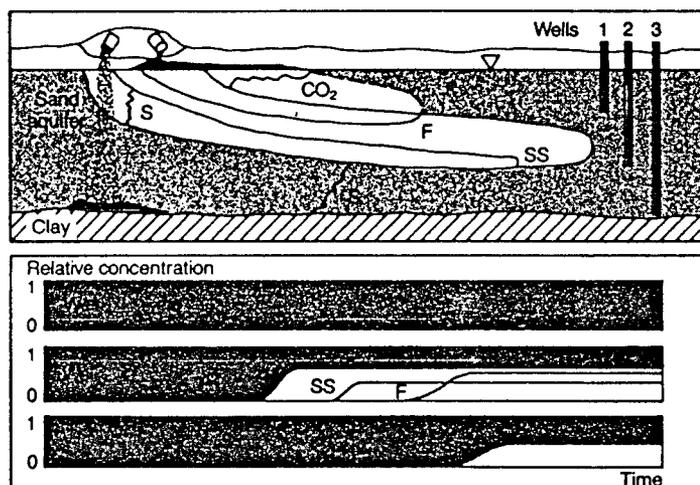
Figure 1c shows the overlapping plumes that might result. It also shows that the dissolving contaminant might affect both of the shallow wells, depending on the size of the source and the degree to which the organic liquid spreads as it floats on the water table. The dissolved contaminant F is assumed to be slightly retarded during transport in the saturated zone. The concentration of F observed in the wells is likely to be considerably lower than its solubility in water as a result of dispersion and sorption.

High-density organic liquids

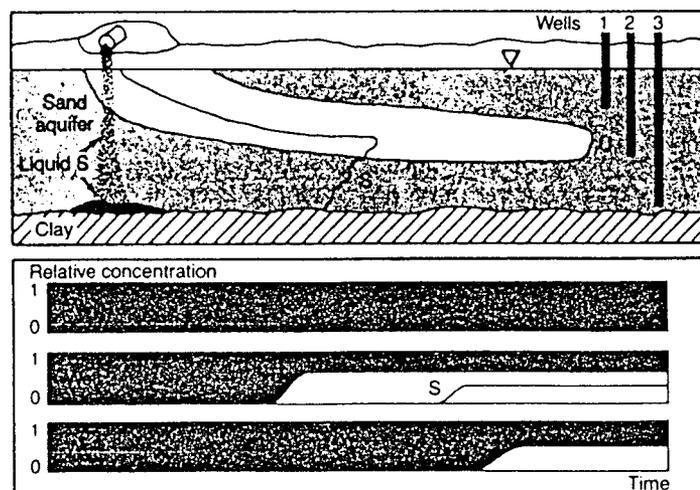
The next case is similar to that just described, except that a small source within the bulk waste releases an or-

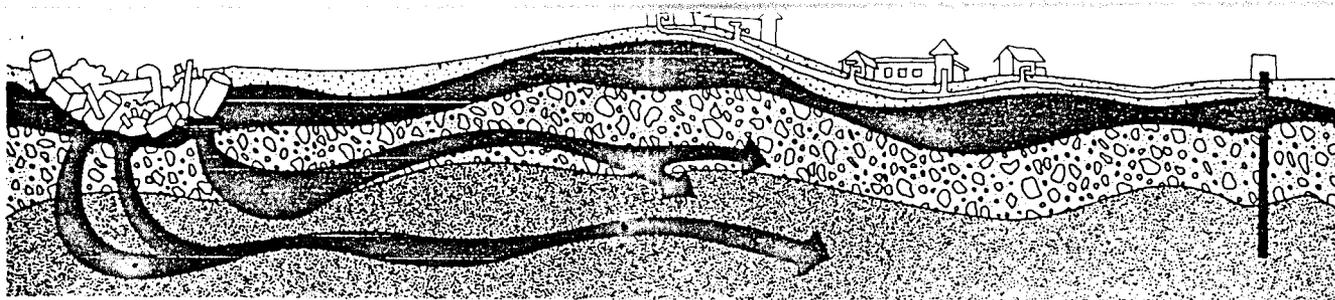
FIGURE 1 (continued)

(d) Continuous source of a dissolved, unretarded contaminant (U) and a pulse source of an organic liquid (S) that sinks through and slowly dissolves into the groundwater



(e) Continuous source of dissolved, unretarded solutes (U) and two pulse sources: floater (F) and sinker (S). The aquifer is aerobic except within the anaerobic U plume. Dissolved F degrades aerobically to CO₂ and water. Dissolved S degrades anaerobically to by-product SS, which is not readily degraded and is less retarded than S or F





ganic liquid contaminant S (e.g., TCE), which is immiscible with and more dense than the groundwater. Liquid S is assumed to sink through the saturated zone as an immiscible phase, displacing the groundwater as it descends, as shown in Figure 1d. Some of the organic liquid is retained by capillary forces in the pores of the aquifer material. If the contaminant is slightly soluble in water, a plume develops by dissolution of the contaminant liquid retained in the aquifer pores as well as by dissolution of the pool of contaminant liquid residing on the bottom of the aquifer.

The resulting plume shape may be complex, depending on the speed of the organic liquid's descent and the amount retained in the aquifer pores. Note that the clay aquitard is assumed level and impervious to the contaminant liquid. Thus, there is no migration of the contaminant pool along the aquifer other than spreading. Figure 1d shows that the two deeper wells will be affected by the contaminant. Also note that the observed concentrations will rise to some value well below the contaminant's solubility. In addition, the contaminant concentration may remain relatively constant for long periods, even when the volume spilled does not seem particularly large from a practical standpoint (a few hundred or thousand gallons of slightly soluble organic liquid contaminants).

Transformable compounds

In a somewhat more complicated case, consider the simple sand-gravel hydrogeologic setting affected by several organic contaminants with different potentials for biotransformation. Assume the leachate from the bulk waste contains readily biodegradable organic solutes. Assume also that the resultant microbiological growth in the aquifer renders the traversed zone anaerobic, as shown by the nonsorbing tracer U in Figure 1e. The remainder of the saturated zone stays aerobic. A small source releases a low-density organic liquid F, which is slightly soluble. Once contaminant F is dissolved, it can be completely mineralized (transformed to carbon dioxide) under aerobic conditions, but it is not biotrans-

formable under anaerobic conditions. Another small source releases an organic liquid "sinker" S, which is slightly soluble. The dissolved contaminant S is not transformable aerobically, but can be biotransformed under anaerobic conditions to an intermediate SS, which is more mobile than S and not readily degradable (for example, TCE transformed anaerobically to vinyl chloride).

In this case, contaminant F disappears from the aerobic zone above the leachate plume but persists within the plume. Conversely, contaminant S persists below the bulk leachate plume but disappears within the plume. Contaminant SS, which would not be found in the waste source itself, appears within the bulk waste leachate plume as a result of biotransformation. In this simplified portrayal no contaminants are observed in Well 1. Well 2 is affected by the bulk leachate plume U, the persistent portion of the F plume, and the transformation product SS. Well 3 is affected by contaminant S only.

Transport time scales

To illustrate the phenomena in a relatively clear and general fashion, the schematic diagrams in Figure 1 do not show specific time and distance scales. In fact, it is difficult to provide such scales with certainty because of the gaps in our understanding of the fundamental processes as they operate under natural conditions. However, if we take a situation that is sufficiently idealized to allow the application of a simple transport model, we can create and examine a quantitative illustration of the effects of advection, dispersion, and sorption.

We assume a sandy aquifer in which the average groundwater flow rate is 45 m/y, within the velocity range that is common. The aquifer is horizontal, unconfined, and has a 10-m-thick saturated zone as indicated in Figure 2a. There is an unlined waste burial pit that is 50 m wide (crossgradient) and a solvent tank buried in a 5-m-wide unlined vault immediately adjacent to it. The bottom of each (the aquifer and the pit) is very near the water table. The waste pit contains chloride ions (Cl^{2-}), TCE, and PCE, which leach continuously.

The leachate contaminates the entire 10-m depth of the aquifer below the impoundment and results in an essentially constant and vertically uniform concentration of the contaminants at $x = 0$. We will assume, for simplicity, that the concentration of each is 1 ppm (1000 ppb) at $x = 0$; this is less than 0.1% and 1% of the solubilities of TCE and PCE, respectively.

We also will assume that simultaneous with the onset of contamination from the waste pit, a small spill occurs as the solvent tank is being filled, and 15 gal (57 L) of pure TCA sinks rapidly through the bottom of the vault, uniformly contaminating the saturated zone in an area 5 m wide. The liquid droplets of TCA in the aquifer then dissolve into the groundwater flowing by, and the TCA concentration reaches 500 ppm, approximately one-half its solubility, before dispersion and dilution begin beyond $x = 0$. Given the volume spilled, the area contaminated by the organic liquid, the initial concentration, the groundwater flow rate, and other typical aquifer characteristics, we calculate that complete dissolution of the TCA droplets takes approximately 10 weeks. It is worth noting, however, that dissolution times on the order of years can be calculated for industrial spills that would still be considered small (hundreds or thousands of gallons).

The dissolved contaminants migrate toward a monitoring well directly downgradient at a distance of 1000 m. Pumping the well during sampling is assumed to have a negligible effect on the gradient. We assume moderate dispersive characteristics for such migration distances in sandy aquifers: 10 m and 1 m for longitudinal and transverse horizontal dispersivity, respectively (3). Also we assume that the chloride ion is not retarded and the retardation factors for TCE, TCA, and PCE are 2, 2, and 3, respectively. Such retardation factors would be expected for transport of these solutes in a sandy aquifer of about 0.1% carbon content (15).

In this highly idealized case, we would expect the water quality history at the monitoring well to be that shown in Figure 2b. The vertical dashed line at about 22 years denotes the solute arrival time expected solely on the basis

of the average groundwater flow rate. However, the chloride concentration rises above a few ppb about five years earlier, owing to longitudinal dispersion. The chloride concentration rises slowly over 10–15 years to a plateau that is only 40–50% of the initial leachate concentration, because of horizontal transverse dispersion.

Figure 2b shows that their retardation causes TCE and PCE to arrive at later times. However, dispersion of the retarded contaminants results in considerable spreading of their fronts and their arrival at the well in significant concentrations 10 to 20 years before the average arrival times expected solely on the basis of the average flow rate and retardation factors (dashed lines at 44 and 66 years).

The combined effect of dispersion and sorption on the TCA pulse is dramatic. The maximum concentration observed at the well is approximately a factor of 2000 below the initial dissolved concentration in the spill zone. In addition, the dissolved TCA pulse, which resulted from 10 weeks' dissolution of the liquid TCA introduced into the aquifer, is detectable (above a few ppb) at the monitoring well for 20 to 30 years.

Because this example is fictitious, the results are useful only as illustrations of possible travel times, degrees of spreading, and so on. Nevertheless, several important implications are clear. Relatively small amounts of contaminant (tens of gallons of pure sol-

vent) can be quite significant. The EPA has proposed a recommended limit of 200 ppb for TCA in drinking water. In this example a 15-gal spill would result in this limit (the horizontal line in Figure 2b) being exceeded for several years at the monitoring well. This example predicts that if only one such spill occurs every few years, the limit would be exceeded continuously once the first pulse of TCA reaches the well because the dispersing pulses would overlap.

Arrival times of retarded contaminants also cannot be estimated safely using only the average groundwater velocity and the retardation factor. This is particularly true for contaminants that cause concern at very low concentrations, such as TCE and PCE. If we assume a maximum acceptable concentration of 10–50 ppb, which is in the range currently under discussion, TCE and PCE would arrive at the monitoring well in our example about 9 and 13 years, respectively, before the expected average times. The arrivals would be even earlier if the aquifer's dispersive characteristics were greater than the rather moderate levels assumed.

Implications

It is clear that the appearance of pollutants in supply wells often follows their release to the subsurface by years, if not decades. Given the rapid expansion in the manufacture and use of synthetic organic chemicals since 1940, such time lags imply that contaminated

wells will continue to be discovered at an increasing rate, even if improved handling and disposal practices prevent any new sources of contamination. Similarly, the onset of contamination at a supply well may mark the front of a set of overlapping plumes of different compounds advancing at different rates, which may affect the well in sequence for decades even if the original contaminant source is removed.

It is therefore imperative that we continue to seek greater understanding of the processes that control the transport and transformation of groundwater contaminants. As this knowledge grows, so will our ability to assess the present and future risks associated with groundwater contamination. Our ability to develop and choose wisely among alternatives for remediation will also increase.

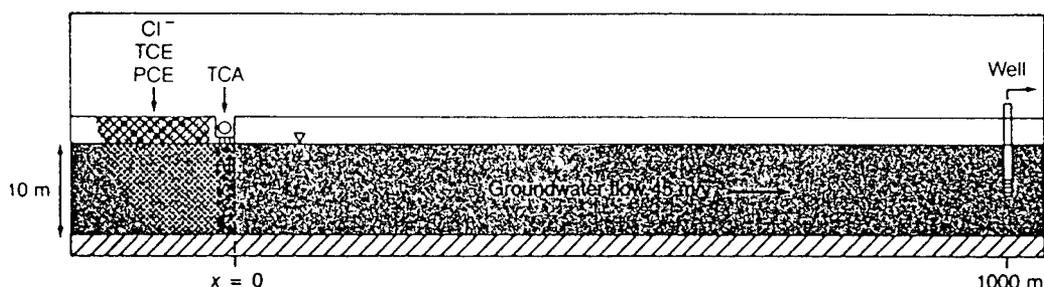
In the following sections, some implications of our current understanding are discussed, highlighting areas of uncertainty to which research resources should be brought to bear.

Monitoring of complex sources

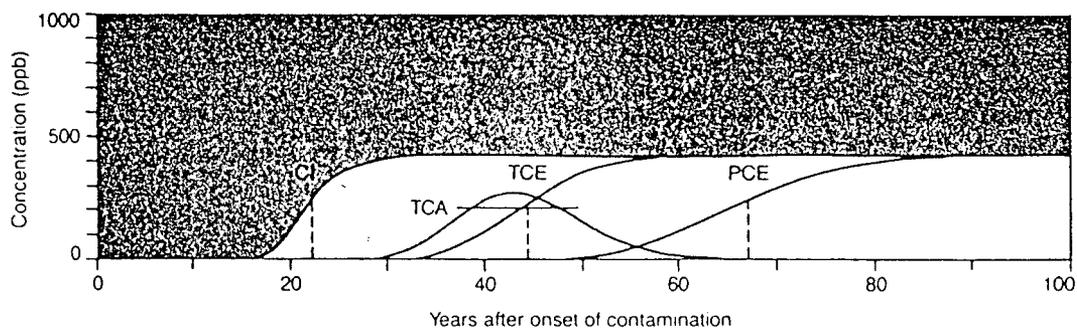
Consideration of the cases presented leads to the conclusion that the distribution of groundwater contaminants at waste disposal or industrial sites is likely to be complex. In each year of operation, landfills used for hazardous waste disposal are certain to have received hundreds or, more likely, thousands of small liquid sources (drums or tank-truck discharges) as well as thou-

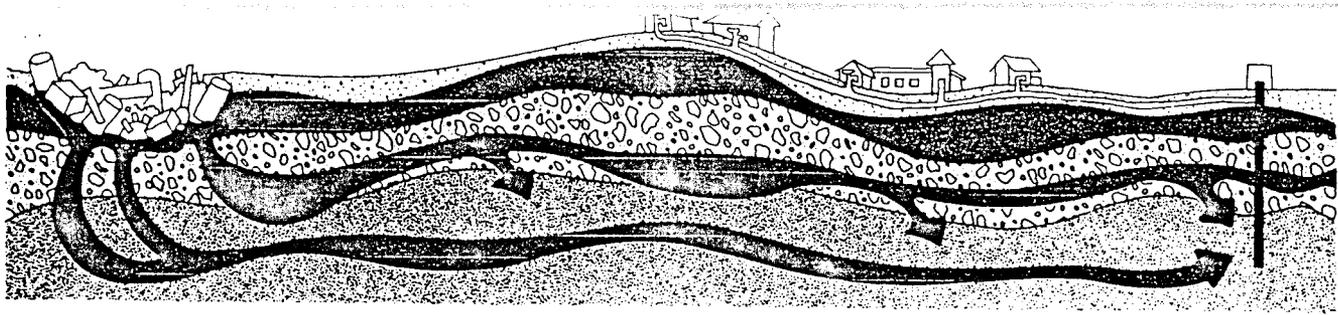
FIGURE 2
Contaminant transport in an idealized aquifer

(a) Schematic of aquifer, contaminant sources (waste pit and buried solvent tank), and well



(b) Concentrations observed at well over time from contaminant releases





sands of loads of solid and semisolid wastes. It is therefore obvious that a given landfill will by no means contain a homogeneous mass of material.

Considering the areal extent of most landfills (from 5 to 200 hectares), the years elapsed during their filling, and the effect of the processes discussed above, it should be expected that the zone of subsurface contamination near unlined landfills on sandy aquifers cannot be delineated as a single plume. Rather the zone must be regarded as a multiplicity of plumes, superposed in three dimensions. Recent field studies support this contention and illustrate the error of assuming that the extent of organic contamination is enveloped by that of noninteractive tracers such as chloride (26). The distribution of subsurface contamination at industrial manufacturing sites also may be complex owing to a variety of contaminant sources such as dry wells, leaking pipes or storage tanks, surface impoundments, and inadvertent spills during chemical handling.

To predict the effect of future contamination near such complex sites, it would be ideal to have a detailed description of each source including the mass released, volume released, and period and area of release. More commonly, the available information is a limited set of observed pollutant concentrations at relatively few monitoring wells, often acquired years after release begins. Thus, the distribution must be deduced from the available data, which may have been obtained at different times with different sampling and analytical methods. Figures 1a-1e illustrate that if the number of reliable observations is limited, there is a high potential for misunderstanding the true contaminant distribution.

The problem is magnified if some or all of the observations from monitoring wells are misleading. The difficulties in obtaining useful and representative measurements of contaminant concentration and distribution have been discussed by a number of authors (2, 27-29). Such difficulties include wells located outside sample contaminant pathways (for example, Well 1 in Figures 1a, 1b, and 1d), wells that sample large depth intervals when contami-

nants are migrating in narrow vertical zones, and wells constructed of materials that alter the quality of the water being sampled. Sample collection, storage, and handling techniques that alter the quality of the sample and inappropriate or incomplete analytical methods also can give misleading information.

Predicting contaminant migration

In addition to descriptions of the contaminant sources, prediction of contaminant migration in the saturated zone requires quantitative representations of advection, dispersion, sorption, and transformation that are specific or at least applicable to the site, contaminants, and period of time in question. Available generalized models for the latter three processes have not yet been convincingly validated using field-scale observations or experiments in even simple sand and gravel aquifers. We must therefore acknowledge that predictions based on them will be uncertain to a degree that may be difficult to define. There is clearly a need for continued research to formulate improved process understanding in models that can be tested at both the laboratory and field scale.

Of particular interest are the interactions among different contaminants in superposed plumes, the rates and products of biotransformation, the effects of complex contaminant distributions on the activity of microorganisms, and the flow and dissolution of immiscible organic liquids. In addition, more resources must be applied to the understanding of contaminant transport and fate in the vadose zone and in heterogeneous hydrogeologic systems, such as discontinuous and interbedded layers of different geologic media and media that contain fractures caused by weathering or seismic activity.

We also need more information about the effects of human activities on the ability of aquitards to prevent vertical flow of contaminants to underlying aquifers. For example, there is evidence from laboratory studies that certain organic liquids can cause desiccation and cracking of unweathered clay, which lead to significant increases in permeability (30, 31). The importance of this effect in the natural environment is un-

known. A more direct effect may be caused by abandoned or improperly installed wells that penetrate aquitards and provide rapid routes to the underlying aquifers. In Santa Clara (Silicon) Valley, Calif., for example, past agricultural activity has left a legacy to the current residential and high-technology industrial cohabitants of perhaps as many as 10,000 inactive and abandoned irrigation wells, the locations of which are largely unknown (32).

Evaluating remedial action

Remedial schemes designed to stop or reverse the spread of groundwater contaminants often rely on pumping the contaminated zone to purge it of contaminants (33). However, if the hydrogeologic situation and the processes that govern the transport of contaminants under natural groundwater flow are not well understood for a specific site, it may be difficult to predict with much confidence what will result from altering natural flow by use of groundwater extraction wells. The processes that disperse and retard the transport of the contaminants can be expected to increase significantly the time and volume of pumping required to decontaminate an area compared with what would be expected based solely on the aquifer volume occupied by the contaminant. The existence of even small volumes of immiscible organic liquids in the subsurface will provide reservoirs of the contaminant and perhaps greatly prolong the pumping time compared with that anticipated when assuming that only dissolved contaminants are present.

Although additional research on transport processes will certainly improve our ability to assess the probable effectiveness of remedial methods, current understanding seems to suggest that remediation based solely on pumping is likely to be a long and expensive undertaking. Pumping is an energy-intensive process and the extracted water may require costly treatment prior to discharge.

These concerns have led to the proposal and investigation of remedial schemes based on in situ biotransformation of contaminants, which may be relatively rapid and inexpensive com-

pared with years of pumping and treatment (21). Although the prospects appear encouraging for remediation by stimulating the activity of microorganisms native to soils and aquifers, much additional laboratory and field research is needed before these techniques can be applied on a practical and significant scale (21).

Conclusion

Understanding of contaminant transport in groundwater has improved greatly in recent years, especially since 1975, as public awareness has grown. We are acquiring a better grasp of the governing principles and have improved our ability to simulate, and in a very limited sense to predict, the movement of some classes of contaminants. Nevertheless, the problems at hand are complex, and there is clearly a need for much more detailed knowledge of the physical, chemical, and microbiological processes that control the fate and transport of contaminants in the subsurface environment.

Applying our conceptual understanding in a general fashion to typical hydrogeologic settings does help explain one perplexing aspect of hazardous chemical contamination: the time sequence of its development and recognition. The extreme time lags that characterize contaminant transport suggest that groundwater contamination by hazardous chemicals is a long-term problem that can reach large proportions before being recognized. It is a problem that is likely to persist long after serious mitigation efforts have begun. It is important to recognize that the time constants for water quality changes in groundwater are large, approaching those that characterize oceans and large lakes and much greater than those for streams.

These long transport times, typically measured in decades, are similar in magnitude to latency periods characterizing chronic health effects such as environmentally induced cancer. This sequence complicates the assessment of the potential for health problems that result from hazardous chemical contamination of subsurface drinking water supplies. Quantitation of health effects based on past exposures may be biased intrinsically toward underestimating the ultimate magnitude of the problem. One thing is certain, groundwater contamination will remain with us for many decades, even if efforts to control present and future emissions are largely successful.

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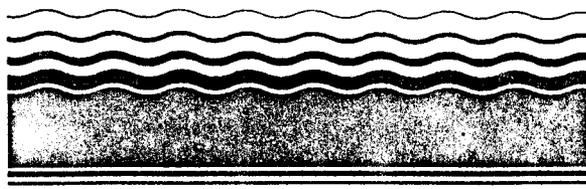


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Technology Demonstration Summary

Terra Vac In Situ Vacuum Extraction System Groveland, Massachusetts

Terra Vac Inc's vacuum extraction system was demonstrated at the Valley Manufactured Products Company, Inc., site in Groveland, Massachusetts. The property is part of the Groveland Wells Superfund site and is contaminated mainly by trichloroethylene (TCE). Vacuum extraction entails removal and venting of volatile organic constituents (VOCs) such as TCE from the vadose or unsaturated zone in the ground by use of extraction wells and vacuum pumps. The process of removing VOCs from the vadose zone using vacuum is a patented process.

The eight-week test run produced the following results:

- extraction of 1,300 lb of VOCs
- a steady decline in the VOC recovery rate with time
- a marked reduction in soil VOC concentration in the test area
- an indication that the process can remove VOCs from clay strata

This Summary was developed by EPA's Risk Reduction Engineering Laboratory, Cincinnati, OH, to announce key findings of the SITE

program demonstration that is fully documented in two separate reports of the same title (see ordering information at back).

Introduction

Environmental regulations enacted in 1984 (and recent amendments to the Superfund program) discourage the continued use of landfilling of wastes in favor of remedial methods that will treat or destroy the wastes. The Superfund program now requires that, to the maximum extent practicable, cleanups at Superfund sites must employ permanent solutions to the waste problem.

The Superfund Innovative Technology Evaluation (SITE) program is one major response to the challenge of finding safe ways to deal with waste sites. Part of the program includes carefully planned demonstration projects at certain Superfund sites to test new waste treatment technologies. These new alternative technologies will destroy, stabilize, or treat hazardous wastes by changing their chemical, biological, or physical characteristics.

Under the SITE program, which is sponsored jointly by the USEPA Office of Research and Development (ORD) and the Office of Solid Waste and Emergency

Response (OSWER), the USEPA selects 10 or 12 Superfund sites each year at which pilot studies of promising technologies can be conducted. Sites are chosen to match the effectiveness and applicability of a particular technology with specific waste types and local conditions. The pilot studies are carefully monitored by the USEPA. Monitoring and data collection determines how effectively the technology treats the waste, how cost-effectively the technology compares with more traditional approaches, and that the operation can be conducted within all public health and environmental guidelines.

The Groveland Wells site was selected for such a demonstration project for 1987. The site is the location of a machine shop, the Valley Manufactured Products Company, Inc., which employs approximately 25 people and manufactures, among other things, parts for valves. The company has been in business at the site since 1964. As an integral part of its building-wide operation of screw machines, the company has used different types of cutting oils and degreasing solvents, mainly trichloroethylene, tetrachloroethylene, trans-1,2-dichloroethylene, and methylene chloride.

The contamination beneath the shop apparently is caused by a leaking storage tank and by former improper practices in the storage and handling of waste oils and solvents. The contamination plume is moving in a northeasterly direction towards and into the Mill Pond.

The USEPA has been involved since 1983, when the Groveland Wells site was finalized on the National Priorities List. The initial Remedial Investigation (RI) of the Valley property was carried out by the responsible party (RP), Valley Manufactured Products Company, Inc. A supplemental RI was conducted by Valley in the fall/winter of 1987 to determine more completely the full nature of contamination at the Valley site. A source control Feasibility Study was performed by USEPA to evaluate various methods for cleaning up or controlling the remaining contaminants. A Record of Decision (ROD) for the site was signed in October 1988 calling for vacuum extraction and groundwater stripping.

The Terra Vac system is being utilized in many locations across the nation. This report is based on monitoring the Terra Vac patented vacuum extraction process (U.S. Patent Nos. 4593760 and 4660639) at the Groveland Wells site during a four-and-one-half-month field operation period, with emphasis on a 56-day

demonstration test active treatment period. The report interprets results of analyses performed on samples and establishes reliable cost and performance data in order to evaluate the technology's applicability to other sites.

The main objectives of this project were:

- The quantification of the contaminants removed by the process.
- The correlation of the recovery rate of contaminants with time.
- The prediction of operating time required before achieving site remediation.
- The effectiveness of the process in removing contamination from different soil strata.

Approach

The objectives of the project were achieved by following a demonstration test plan, which included a sampling and analytical plan. The sampling and analytical plan contained a quality assurance project plan. This QAPP assured that the data collected during the course of this project would be of adequate quality to support the objectives.

The sampling and analytical program for the test was split up into a pretest period, which has been called a pretreatment period, an active period, midtreatment, and a posttreatment period.

The pretreatment period sampling program consisted of:

- soil boring samples taken with split spoons
- soil boring samples taken with Shelby tubes
- soil gas samples taken with punch bar probes

Soil borings taken by split spoon sampling were analyzed for volatile organic compounds (VOCs) using headspace screening techniques, purge and trap, GC/MS procedures, and the EPA-TCLP procedure. Additional properties of the soil were determined by sampling using a Shelby tube, which was pressed hydraulically into the soil by a drill rig to a total depth of 24 feet. These Shelby tube samples were analyzed to determine physical characteristics of the

subsurface stratigraphy such as bulk density, particle density, porosity, pH, grain size, and moisture. These parameters were used to define the basic soil characteristics.

Shallow soil gas concentrations were collected during pre-, mid-, and post-treatment activities. Four shallow vacuum monitoring wells and twelve shallow punch bar tubes were used at sample locations. The punch bar samples were collected from hollow stainless steel probes that had been driven to a depth of 3 to 5 feet. Soil gas was drawn up the punch bar probes with a low-volume personal pump and tygon tubing. Gas-tight 50-ml syringes were used to collect the sample out of the tygon tubing.

The active treatment period consisted of collecting samples of:

- wellhead gas
- separator outlet gas
- primary carbon outlet gas
- secondary carbon outlet gas
- separator drain water

All samples with the exception of the separator drain water were analyzed on site. On-site gas analysis consisted of gas chromatography with a flame ionization detector (FID) or an electron capture detector (ECD). The FID was used generally to quantify the trichloroethylene (TCE) and trans 1,2-dichloroethylene (DCE) values, while the ECD was used to quantify the 1,1,1-trichloroethane (TRI) and the tetrachloroethylene (PCE) values.

The separator drain water was analyzed for VOC content using SW846 8010. Moisture content of the separator inlet gas from the wells was analyzed using EPA Modified Method 4. This method is good for the two-phase flow regime that existed in the gas emanating from the wellhead. See Table 1 for a listing of analytical methods applied.

The posttreatment sampling essentially consisted of repeating pretreatment sampling procedures at locations as close as possible to the pretreatment sampling locations.

The activated carbon canisters were sampled, as close to the center of the canister as possible, and these samples were analyzed for VOC content as a check on the material balance for the process. The method used was P&CAM 127, which consisted of desorption of the carbon with CS₂ and subsequent gas chromatographic analysis.

Table 1. Analytical Methods

<i>Parameter</i>	<i>Analytical Method</i>	<i>Sample Source</i>
Grain size	ASTM D422-63	Soil borings
pH	SW846* 9040	Soil borings
Moisture (110°C)	ASTM D2216-80	Soil borings
Particle density	ASTM D698-78	Soil borings
Oil and grease	SW846* 9071	Soil borings
EPA-TCLP	F. R. 11/7/86, Vol. 51, No. 216, SW846* 8240	Soil borings
TOC	SW846* 9060	Soil borings
Headspace VOC	SW846* 3810	Soil borings
VOC	GC/FID or ECD	Soil gas
VOC	GC/FID or ECD	Process gas
VOC	SW846* 8010	Separator liquid
VOC	SW846* 8010	Groundwater
VOC	Modified P&CAM 127	Activated carbon
VOC	SW846* 8240	Soil borings

*Third Edition, November 1986.

Process Description

The vacuum extraction process is a technique for the removal and venting of volatile organic constituents (VOCs) from the vadose or unsaturated zone of soils. Once a contaminated area is completely defined, an extraction well or wells, depending upon the extent of contamination, will be installed. A vacuum system induces air flow through the soil, stripping and volatilizing the VOCs from the soil matrix into the air stream. Liquid water is generally extracted as well along with the contamination. The two-phase flow of contaminated air and water flows to a vapor liquid separator where contaminated water is removed. The contaminated air stream then flows through activated carbon canisters arranged in a parallel-series fashion. Primary or main adsorbing canisters are followed by a secondary or backup adsorber in order to ensure that no contamination reaches the atmosphere.

Equipment Layout and Specifications

The equipment layout is shown in Figure 1, and specifications are given in Table 2 for the equipment used in the initial phase of the demonstration. This equipment was later modified when unforeseen circumstances required a shutdown of the system. The vapor-liquid separator, activated carbon canisters, and vacuum pump skid were inside the building, with the stack discharge outside the building. The equipment was in an

area of the machine shop where used cutting oils and metal shavings had been stored.

Four extraction wells (EW1 - EW4) and four monitoring wells (MW1 - MW4) were drilled south of the shop. Each well was installed in two sections, one section to just above the clay lens and one section to just below the clay lens. The extraction wells were screened above the clay and below the clay. As shown in Figure 2, the well section below the clay lens was isolated from the section above by a bentonite portland cement grout seal. Each section operated independently of the other. The wells were arranged in a triangular configuration, with three wells on the base of the triangle (EW2, EW3, EW4) and one well at the apex (EW1). The three wells on the base were called barrier wells. Their purpose was to intercept contamination, from underneath the building and to the side of the demonstration area, before this contamination reached the main extraction well (EW1). The area enclosed by the four extraction wells defined the area to be cleaned.

Installation of Equipment

Well drilling and equipment setup were begun on December 1, 1987. A mobile drill rig was brought in and equipped with hollow-stem augers, split spoons, and Shelby tubes. The locations of the extraction wells and monitoring wells had been staked out based on contaminant concentration profiles from a previously

conducted remedial investigation and from bar punch probe soil gas monitoring.

Each well drilled was sampled at 2-foot intervals with a split spoon pounded into the subsurface by the drill rig in advance of the hollow stem auger. The hollow stem auger would then clear out the soil down to the depth of the split spoon, and the cycle would continue in that manner to a depth of 24 feet. The drilling tailings were shoveled into 55-gallon drums for eventual disposal. After the holes were sampled, the wells were installed using 2-inch PVC pipes screened at various depths depending upon the characteristics of the soil in the particular hole. The deep well was installed first, screened from the bottom to various depths. A layer of sand followed by a layer of bentonite and finally a thick layer of grout were required to seal off the section below the clay lens from the section above the clay lens. The grout was allowed to set overnight before the shallow well pipe was installed at the top of the grout. A layer of sand bentonite and grout finished the installation.

VOC Removal From the Vadose Zone

The permeable vadose zone at the Groveland site is divided into two layers by a horizontal clay lens, which is relatively impermeable. As explained previously, each extraction well had a separate shallow and deep section to enable VOCs to be extracted from that

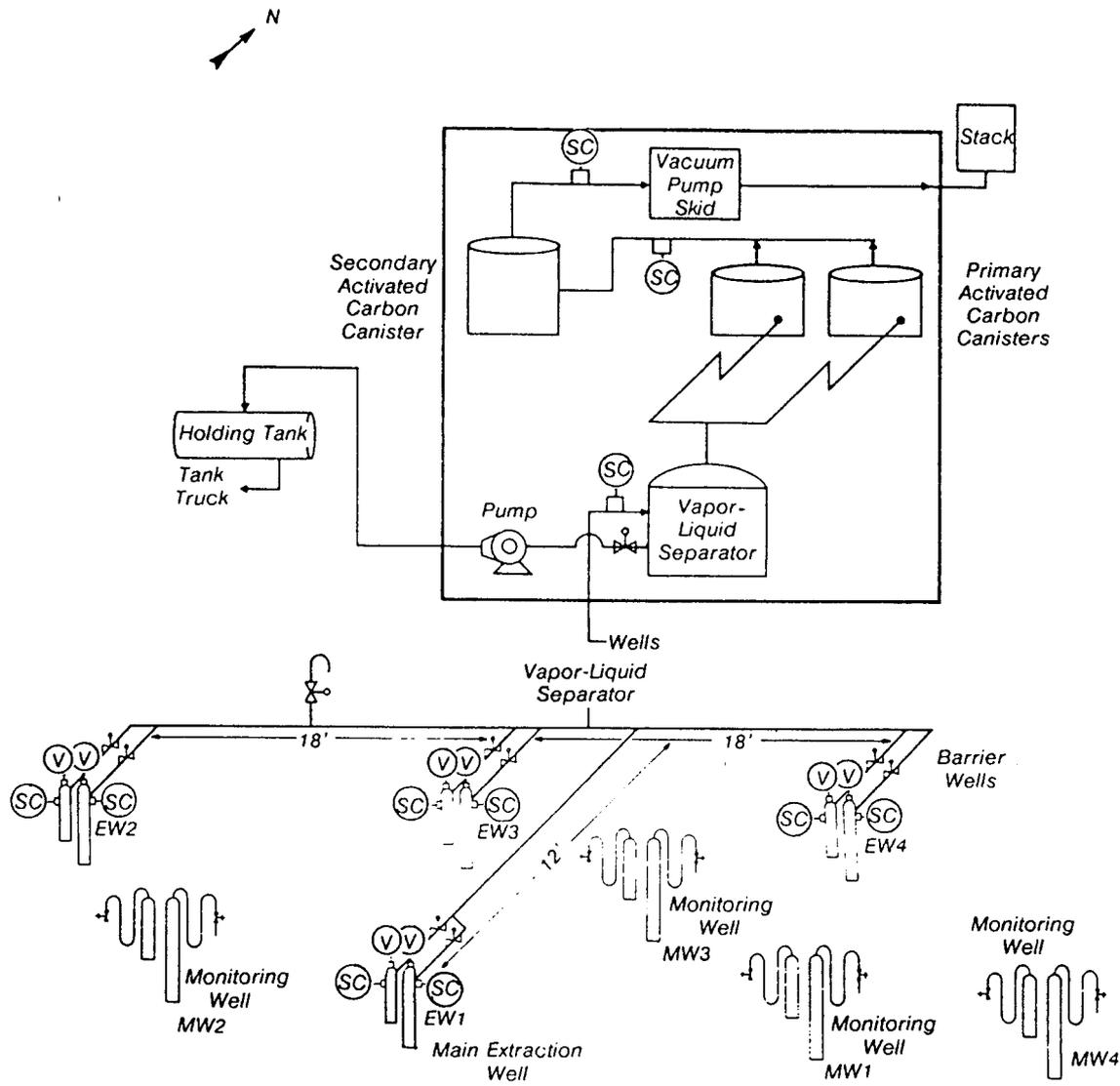


Figure 1. Schematic diagram of equipment layout.

Table 2. Equipment List

Equipment	Number Required	Description
Extraction wells	4 (2 sections each)	2" SCH 40 PVC 24' total depth
Monitoring wells	4 (2 sections each)	2" SCH 40 PVC 24' total depth
Vapor-liquid separator	1	1000-gal capacity, steel
Activated carbon canisters	Primary: 2 units in parallel Secondary: 1 unit	Canisters with 1200 lb of carbon in each canister - 304 SS 4" inlet and outlet nozzles
Vacuum unit	1	Terra Vac Recovery Unit - Model PR17 (25 HP Motor)
Holding tank	1	2000-gal capacity - steel
Pump	1	1 HP motor - centrifugal

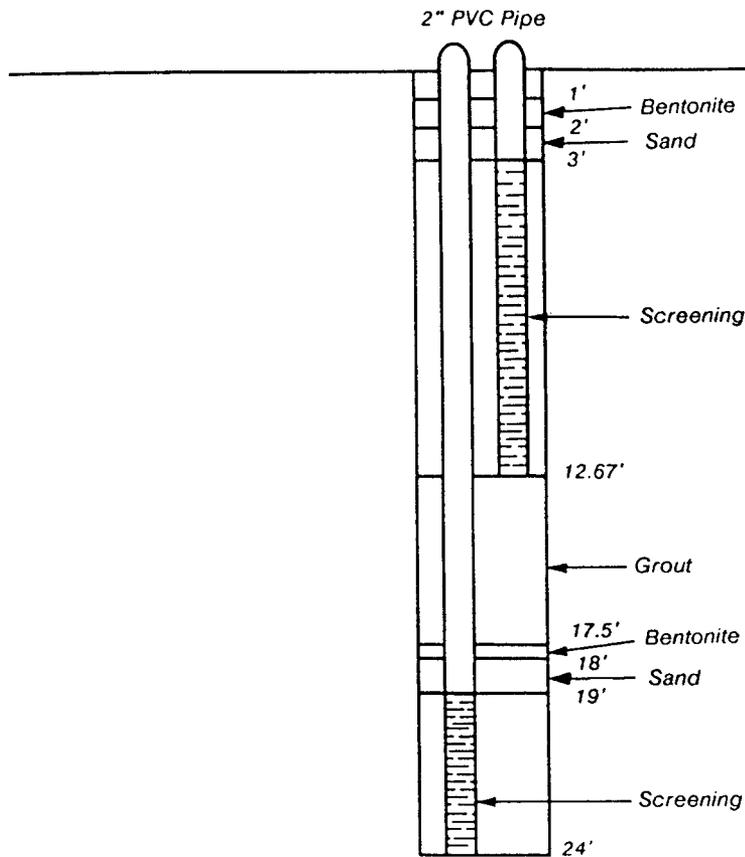


Figure 2. Schematic diagram of an extraction well.

area of the vadose zone above and below the clay lens. The quantification of VOCs removed was achieved by measuring

- gas volumetric flow rate by rotameter and wellhead gas VOC concentration by gas chromatography
- the amount of VOCs adsorbed by the activated carbon canisters by desorption into CS₂ followed by gas chromatography.

VOC flow rates were measured and tabulated for each well section separately. The results of gas sampling by syringe and gas chromatographic analysis indicate a total of 1,297 lb of VOCs were extracted over a 56-day period, 95% of which was trichloroethylene. A very good check on this total was made by the activated carbon VOC analysis, the results of which indicated a VOC recovery of 1353 lb; virtually the same result was obtained by two very different methods.

The soil gas results show a considerable reduction in concentration over the course of the 56-day demonstration period as can be seen from Figures 3 and 4. This is to be expected since soil gas is the vapor halo existing around the contamination and should be relatively easy to remove by vacuum methods.

A more modest reduction can be seen in the results obtained for soil VOC concentrations by GC/MS purge-and-trap analytical techniques. Soil concentrations include not only the vapor halo but also interstitial liquid contamination that is either dissolved in the moisture in the soil or exists as a two-phase liquid with the moisture.

Table 3 shows the reduction of the weighted average TCE levels in the soil during the course of the 56-day demonstration test. The weighted average TCE level was obtained by averaging soil concentrations obtained every two feet by split spoon sampling methods over the entire 24-foot depth of the wells. The largest reduction in soil TCE concentration occurred in extraction

well 4, which had the highest initial level of contamination. Extraction well 1, which was expected to have the greatest concentration reduction potential, exhibited only a minor decrease over the course of the test. Undoubtedly this was because of the greater-than-expected level of contamination that existed in the area around monitoring well 3 that was drawn into the soil around extraction well 1. The decrease in the TCE level around monitoring well 3 tends to bear this out.

Effectiveness of the Technology in Various Soil Types

The soil strata at the Groveland site can be characterized generally as consisting of the following types in order of increasing depth to groundwater:

- medium to very fine silty sands
- stiff and wet clays
- sand and gravel

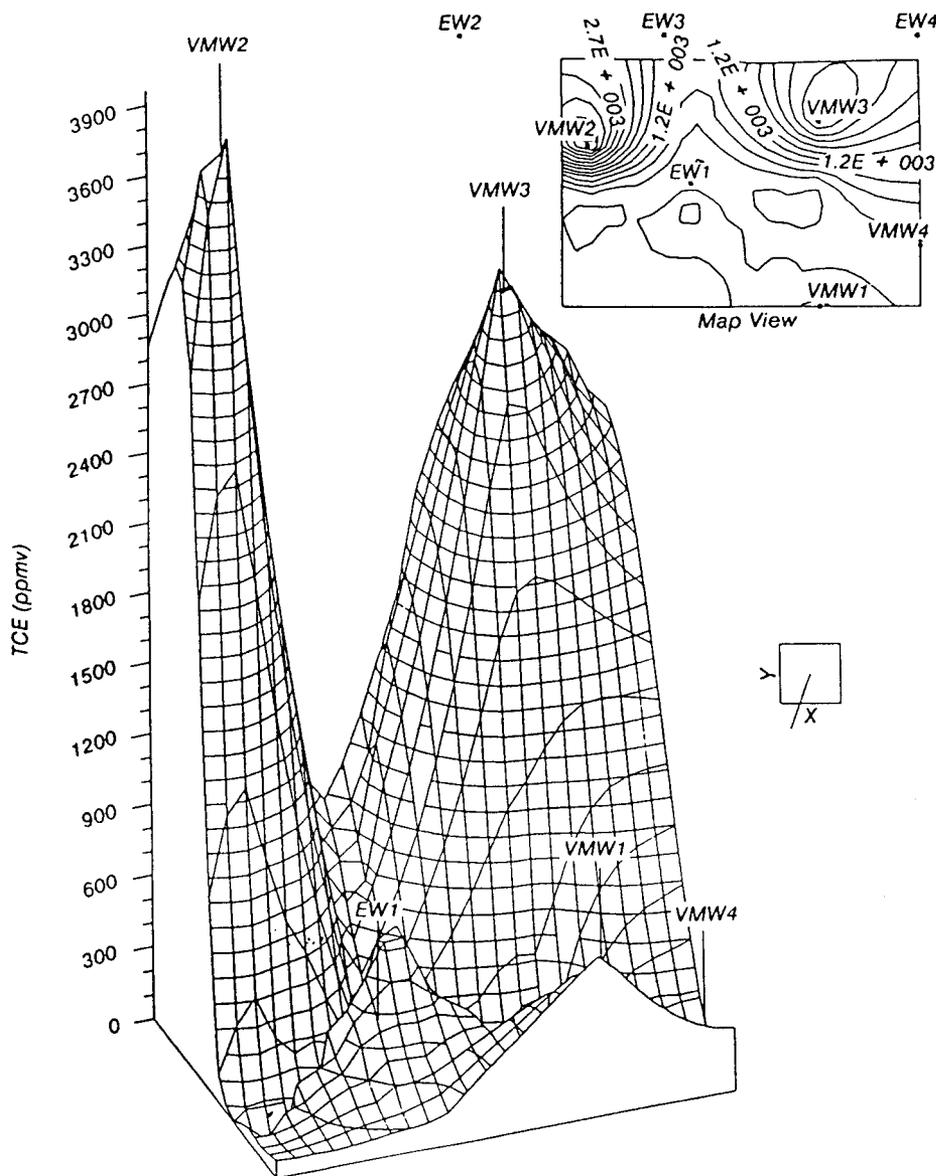


Figure 3. Pretreatment shallow soil gas concentration.

Soil porosity, which is the percentage of total soil volume occupied by pores, was relatively the same for both the clays and the sands. Typically porosity, over the 24-foot depth of the wells, would range between 40% and 50%. Permeabilities, or more accurately hydraulic conductivities, ranged from 10^{-4} cm/sec for the sands to 10^{-8} cm/sec for the clays with corresponding grain sizes equal to 10^{-1} mm to 10^{-3} mm.

Pretest soil boring analyses indicated in general that most of the contamination was in the strata above the clay lens, with a considerable quantity perched on top of the clay lens. This was the case for ex-

traction well 4, which showed an excellent reduction of TCE concentration in the medium to fine sandy soils existing above the clay layer, with no TCE detected in the clay in either the pretest or posttest borings (see Table 4). One of the wells, however, was an exception. This was monitoring well 3, which contained the highest contamination levels of any of the wells, and was exceptional in that most of the contamination was in a wet clay stratum. The levels of contamination were in the 200 to 1600 ppm range before the test. After the test, analyses of the soil boring adjacent to monitoring well 3 showed levels in the

range of ND-60 ppm in the same clay stratum. The data suggest that the technology can desorb or otherwise mobilize VOCs out of certain clays (see Table 5).

From the results of this demonstration it appears that the permeability of a soil need not be a consideration in applying the vacuum extraction technology. This may be explained by the fact that the porosities were approximately the same for all soil strata, so that the total flow area for stripping air was the same in all soil strata. It will take a long time for a liquid contaminant to percolate through clay with its small pore size and

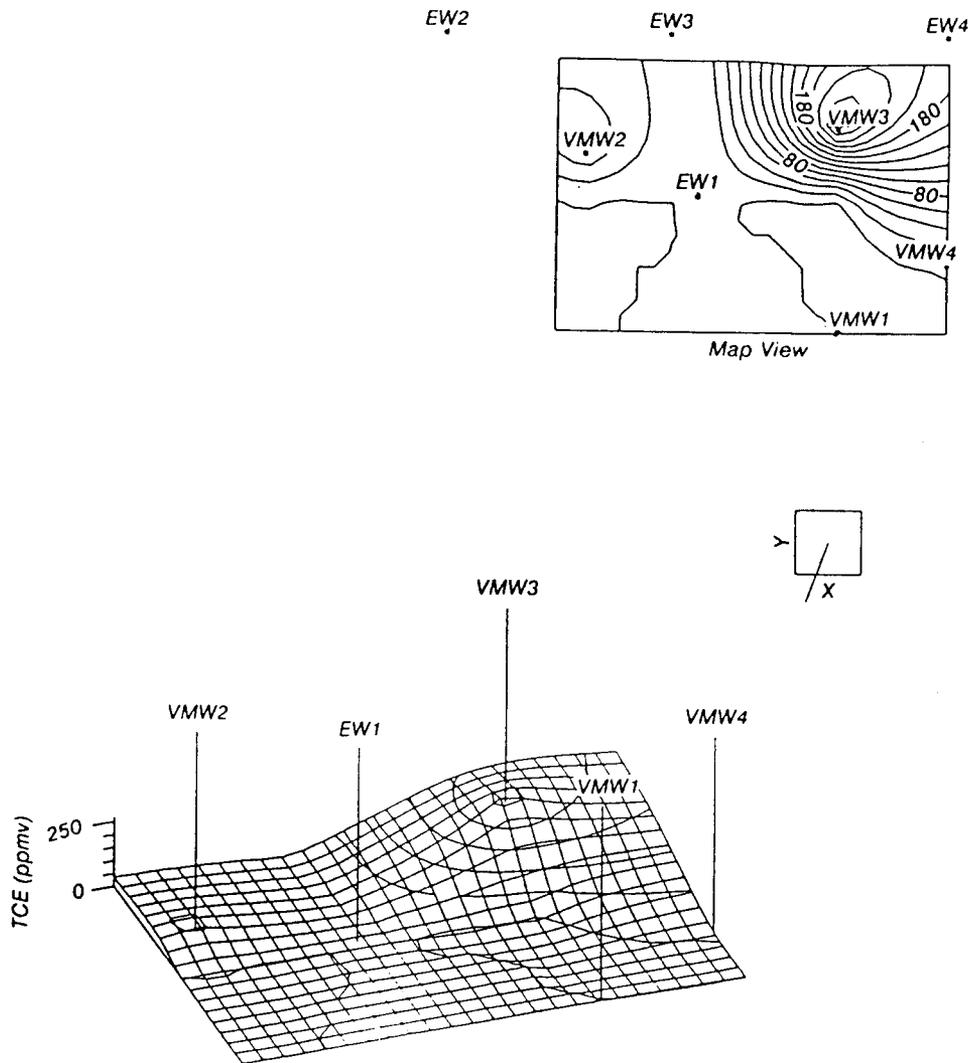


Figure 4. Posttreatment shallow soil gas concentration.

consequent low permeability. However, the much smaller air molecules have a lower resistance in passing through the same pores. This may explain why contamination was generally not present in the clay strata but when it was, it was not difficult to remove. Further testing should be done in order to confirm this finding.

Correlation of Declining VOC Recovery Rates

The vacuum extraction of volatile organic constituents from the soil may be

viewed as an unsteady state process taking place in a nonhomogeneous environment acted upon by the combined convective forces of induced stripping air and by the vacuum induced volatilization and diffusion of volatiles from a dissolved or sorbed state. As such it is a very complicated process, even though the equipment required to operate the process is very simple.

Unsteady state diffusion processes in general correlate well by plotting the logarithm of the rate of diffusion versus

time. Although the representation of the vacuum extraction process presented here might be somewhat simplistic, the correlation obtained by plotting the logarithm of the concentration of contaminant in the wellhead gas versus time and obtaining a least squares best fit line was reasonably good. This type of plot, shown in Figure 5, represents the data very well and is more valid than both a linear graph or one plotting concentration versus log time, in which a best fit curve would actually predict gas concentrations of zero or less.

Table 3. Reduction of Weighted Average TCE Levels in Soil (TCE Conc. in mg/kg)

Extraction Well	Pretreatment	Posttreatment	% Reduction
1	33.98	29.31	13.74
2	3.38	2.36	30.18
3	6.89	6.30	8.56
4	96.10	4.19	95.64
Monitoring Well			
1	1.10	0.34	69.09
2	14.75	8.98	39.12
3	227.31	84.50	62.83
4	0.87	1.05	--

Table 4. Extraction Well 4—TCE Reduction in Soil Strata

Depth ft	Description of Strata	Perme- ability cm/sec	TCE Conc. ppm	
			pre	post
0-2	Med. sand w/gravel	10 ⁻⁴	2.94	ND
2-4	Lt. brown fine sand	10 ⁻⁴	29.90	ND
4-6	Med. stiff lt. brown fine sand	10 ⁻⁵	260.0	39
6-8	Soft dk. brown fine sand	10 ⁻⁵	303.0	9
8-10	Med. stiff brown sand	10 ⁻⁴	351.0	ND
10-12	V stiff lt. brown med. sand	10 ⁻⁴	195.0	ND
12-14	V stiff brown fine sand w/silt	10 ⁻⁴	3.14	2.3
14-16	M stiff grn-brn clay w/silt	10 ⁻⁸	ND	ND
16-18	Soft wet clay	10 ⁻⁸	ND	ND
18-20	Soft wet clay	10 ⁻⁸	ND	ND
20-22	V stiff brn med-coarse sand	10 ⁻⁴	ND	ND
22-24	V stiff brn med-coarse w/gravel	10 ⁻³	6.71	ND

Table 5. Monitoring Well 3—TCE Reduction in Soil Strata

Depth ft	Description of Strata	Perme- ability cm/sec	TCE Conc. ppm	
			pre	post
0-2	M. stiff brn. fine sand	10 ⁻⁵	10.30	ND
2-4	M. stiff grey fine sand	10 ⁻⁵	8.33	800
4-6	Soft lt. brn. fine sand	10 ⁻⁴	80.0	84
6-8	Lt. brn. fine sand	10 ⁻⁴	160.0	ND
8-10	Stiff V. fine brn. silty sand	10 ⁻⁴	ND	63
10-12			NR	2.3
12-14	Soft brown silt	10 ⁻⁴	316.0	ND
14-16	Wet green-brown silty clay	10 ⁻⁸	195.0	ND
16-18	Wet green-brown silty clay	10 ⁻⁸	218.0	62
18-20	Wet green-brown silty clay	10 ⁻⁸	1570.0	2.4
20-22	Silt, gravel, and rock frag.	10 ⁻⁴	106.0	ND
22-24	M. stiff lt. brn. med. sand	10 ⁻⁴	64.1	ND

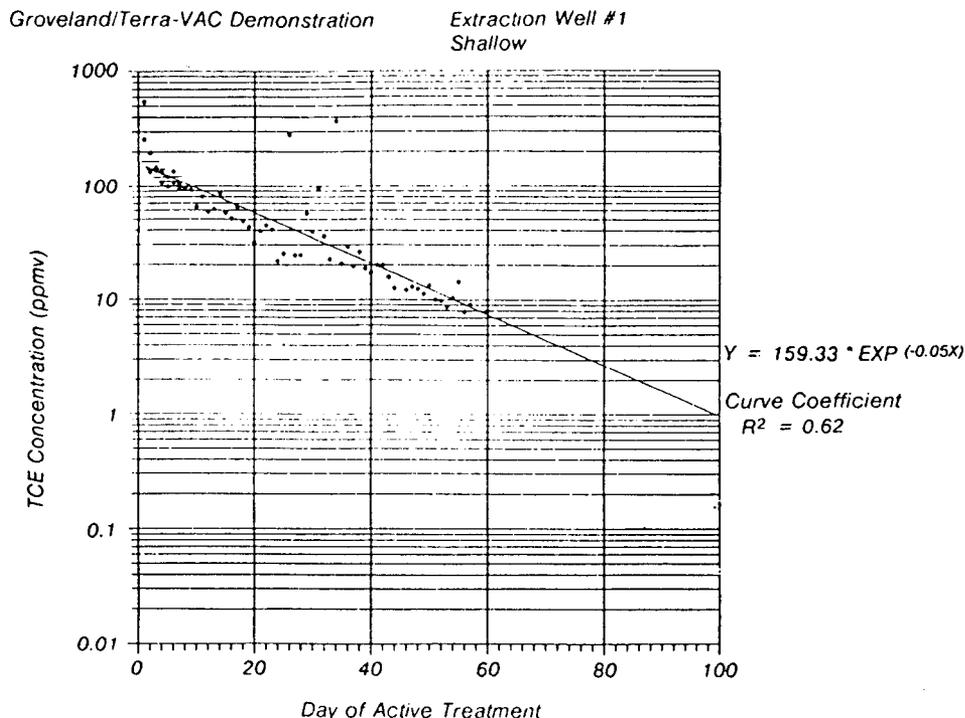


Figure 5. Wellhead TCE concentration vs time.

Looking at the plots for extraction well 1, shallow and deep, equations are given for the least squares best fit line for the data points. If the vacuum extraction process is run long enough to achieve the detection limit for TCE on the ECD, which is 1 ppbv, the length of time required to reach that concentration would be approximately 250 days on the

shallow well and approximately 300 days on the deep well.

Prediction of Time Required for Site Remediation

The soil concentration that would be calculated from the wellhead gas concentration using Henry's Law is in-

cluded in the last column of Table 6. Calculations for the predicted soil concentrations were made assuming a bulk density of the soil of 1761 kg/m³, a total porosity of 50%, and a moisture content of 20%. The calculated air filled porosity of the soil is approximately 15%. Henry's constant was taken to be 0.492 KPa·m³-gmol at 40°F.

Table 6. Comparison of Wellhead Gas VOC Concentration and Soil VOC Concentration

Extraction Well	TCE Concentration in Wellhead Gas ppmv	TCE Concentration in Soil ppmw	Predicted by Henry's Law ppmw
1S	9.7	54.5	0.11
1D	5.6	7.2	0.07
2S	16.4	ND	0.20
2D	14.4	20.4	0.17
3S	125.0	20.9	1.53
3D	58.7	18.0	0.74
4S	1095.6	9.1	12.49

Given the nonhomogeneous nature of the subsurface contamination and interactions of TCE with organic matter in the soil, it was not possible to obtain a good correlation between VOC concentrations in wellhead gas and soil in order to predict site remediation times. Henry's Law constants were used to calculate soil concentrations from wellhead gas concentrations and the calculated values obtained, correcting for air filled porosity, were lower than actual soil concentrations by at least an order of magnitude (see Table 6).

Before one can attempt to make a rough estimation of the remediation time, a target value for the particular contaminant in the remediated soil must be calculated. This target concentration is calculated by using two mathematical models, the Vertical and Horizontal Spread Model (VHS) and the Organic Leachate Model (OLM) (EPA Draft Guidelines for Petitioning Waste Generated by the Petroleum Refinery Industry, June 12, 1987). The mathematical models allow the use of a regulatory standard for drinking water in order to arrive at a target soil concentration.

The VHS model is expressed as the following equation:

$$C_y = C_o \operatorname{erf} (Z/(2(a_2 Y)^{0.5})) \operatorname{erf} (X/(a_1 Y)^{0.5})$$

where:

C_y = concentration of VOC at compliance point (mg/l)

C_o = concentration of VOC in leachate (mg/l)

erf = error function (dimensionless)

Z = penetration depth of leachate into the aquifer

Y = distance from site to compliance point (m)

X = length of site measured perpendicular to the direction of groundwater flow (m)

a_1 = lateral transverse dispersivity (m)

a_2 = vertical dispersivity (m)

A simplified version of the VHS model is most often used, which reduces the above equation to:

$$C_y = C_o C_1$$

where:

$C_1 = \operatorname{erf} (Z/(2(a_2 Y)^{0.5})) \operatorname{erf} (X/(a_1 Y)^{0.5})$, which is reduced to a conversion factor corresponding to the amount of contaminated soil

The Organic Leachate Model (OLM) is written as:

$$C_o = 0.00211 C_s^{0.678} S^{0.373}$$

where:

C_o = concentration of VOC in leachate (mg/l)

C_s = concentration of VOC in soil (mg/l)

S = solubility of VOC in water (mg/l)

The regulatory standard for TCE in drinking water is 3.2 ppb. This regulatory limit is used in the VHS model as the compliance point concentration in order to solve for a value of the leachate concentration. This value of leachate concentration is then used in the OLM model to solve for the target soil concentration.

Once the target soil concentration is determined, a rough estimation of the remediation time can be made by taking the ratio of soil concentration to wellhead gas concentration and extrapolating in order to arrive at a wellhead gas concentration at the target soil concentration. The calculated target soil concentration for this site is 500 ppbw. This corresponds to an approximate wellhead gas concentration of 89 ppb for EW1S. The equation correlating wellhead gas concentration with time (see Figure 5) is then solved to give 150 days running time.

After 150 days the vacuum extraction system can be run intermittently to see if significant increases in gas concentrations occur upon restarting, after at least a two-day stoppage. If there are no appreciable increases in gas concentration, the soil has reached its residual equilibrium contaminant concentration and the system may be stopped and soil borings taken and analyzed.

The full report was submitted in fulfillment of Contract No. 68-03-3255 by Foster Wheeler Enviroresponse, Inc., under the sponsorship of the U.S. Environmental Protection Agency.

The EPA Project Manager, **Mary Stinson**, is with the Risk Reduction Engineering Laboratory, Edison, NJ 08837 (see below).

The complete report consists of two volumes entitled "Technology Evaluation Report: SITE Program Demonstration Test, Terra Vac In Situ Vacuum Extraction System, Groveland, Massachusetts:"

"Volume I" (Order No. PB 89-192 025/AS; Cost: \$21.95, subject to change) discusses the results of the SITE demonstration

"Volume II" (Order No. PB 89-192 033/AS; Cost: \$36.95, subject to change) contains the technical operating data logs, the sampling and analytical data, and the quality assurance data

Both volumes of this report will be available only from:

National Technical Information Service

5285 Port Royal Road

Springfield, VA 22161

Telephone: 703-487-4650

A related report, entitled "Application Analysis Report: Terra Vac In Situ Vacuum Extraction System," which discusses the applications and costs, is under development.

The EPA Project Manager can be contacted at:

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SOIL AND GROUNDWATER REMEDIATION
USING MODIFIED PUMP AND TREAT
AND TWO-PHASE VACUUM EXTRACTION TECHNOLOGIES

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ABSTRACT

Solvent contamination is a common, worldwide problem. Often, these releases contaminate not only the upper unsaturated soils, but extend to the underlying saturated zone and groundwater. The ultimate goals in addressing these environmental liabilities are to effectively remediate the situation, minimize costs, and satisfy regulatory agencies. Two remediation technologies have proven to be cost-effective treatment options for solvent contamination that extends to the water table. These technologies are a modified groundwater pump and treat system with vacuum extraction, and a two-phase vacuum extraction system.

Dames & Moore has successfully used groundwater pump and treat systems in conjunction with vacuum extraction. The vacuum extraction process removes volatile solvents and fuel contamination in the existing unsaturated (vadose) zone and in the vadose zone that is created when the water table falls (cone of depression) during active pumping of the contaminated groundwater.

Two-phase vacuum extraction is a system that allows the removal of vadose-zone contaminants and groundwater with a single vacuum pump. Under applicable conditions, this technology can provide a rapid and cost-effective means of remediation.

These technologies provide in-situ remediation, which minimizes impact on the site and can reduce the time required for cleanup. They have proven to be reliable and cost-effective methods of efficiently removing the vast majority of volatile solvents from the treated area. Because these technologies remove the solvents from the groundwater and soil at the remediated site, potential future concerns of contaminant mobility or recontamination are minimized or eliminated.

These two technologies can be successfully applied to certain contaminant situations. At appropriate sites, the modified pump and treat systems with vacuum extraction and the two-phase vacuum extraction systems offer advantages over other technologies.

INTRODUCTION

Historic fuel and solvent spills have significantly impacted soil and groundwater throughout the United States and the world. Environmental problems are being discovered at industrial and military facilities, even corner mom-and-pop gasoline stores at an ever increasing rate. These spills frequently involve solvents, commonly referred to as volatile organic compounds (VOCs), or fuel oils.

VOC contamination, whether from recent or historic release events, typically emanates from leaking tanks or pipes, or from mishandling or intentional spilling of material. These releases frequently contaminate not only the upper unsaturated soils (the vadose zone), but extend to the underlying saturated zone and groundwater. Proper characterization, delineation, and remediation of these conditions involves substantial costs.

The ultimate goals in addressing these environmental liabilities are to effectively remediate the contamination, minimize remedial costs, and satisfy the regulatory requirements. Although the complete arsenal of available treatment technologies must be evaluated, two technologies have proven to be cost-effective treatment options for sites involving volatile solvent contamination that extends to the water table. These technologies are a modified groundwater pump and treat system involving volatile vacuum extraction (VE), and a two-phase vacuum extraction system.

These technologies are similar and offer numerous advantages over other technologies, including:

- These in-situ processes involve minimal disruption to the facility operations
- VOCs are removed from the vadose zone
- The time required to remediate the site may be lessened

Other advantages of these technologies are discussed later in this paper.

Individual discussions of these two technologies as they relate to VOC contamination are divided into the following items:

- A review of the technology
- The required equipment for the implementation of the technology
- Advantages and disadvantages of the technology
- Costs, reliability, and limitations
- Case histories of the application of the technology and reportable results

Sections discussing the future direction of the technologies and conclusions are presented at the end of this paper.

MODIFIED PUMP AND TREAT SYSTEM WITH VACUUM EXTRACTION

Review of the Technology

The standard pump and treat system, which pumps contaminated groundwater out of the ground for aboveground treatment, is used extensively and has proven to be a solid performer in the arsenal of remedial technologies. However, it can have potential and significant drawbacks, depending on the application, including:

- Inability to remediate the vadose zone contaminants
- Limited ability to remove contaminants remaining in the soil matrix of the newly formed vadose zone, after the surrounding groundwater have been depressed by pumping

Although these limitations can be overcome, the solutions may be undesirable or require additional time or money. For example, clean water can be recharged into the ground to flush contamination from the vadose zone, but this action may be impractical for contamination under a building. Another solution is to cap the affected area to minimize impact without remediating a potential source. VOCs remaining in the newly formed vadose zone can also be addressed by periodically shutting down the pumping system. This shut down allows the groundwater to resaturate the soils and mobilize the residual contaminants into the groundwater for subsequent removal when pumping is reinstated.

These potential limitations to the standard pump and treat system can be minimized or eliminated in many situations by the addition of vacuum extraction. VE technology is a recent addition to the range of remediation options and has proven to be an effective tool against VOC contamination.

VE is normally the application of a vacuum on a well (extraction well), that features a screened interval extending through the zone of soil contamination. The well can be installed vertically or horizontally in a trench. The vacuum lowers the pressure in the soil adjacent to the well, thereby inducing an air flow toward the well. As air is pulled through the soil matrix, VOCs in the vadose zone volatilize into the air stream, which is removed from the ground through the well. Generally, the air stream is treated at the surface to remove the VOCs.

The vapor pressure of a particular compound determines whether vacuum extraction will effectively remove that compound from the soil. In general, compounds with vapor pressure greater than 10 mm mercury (Hg) at 20°C are strong candidates for VE. Light fuels, such as gasoline and kerosene, and many solvents fulfill this requirement.

The radius of influence (ROI) of an extraction well will depend on the type(s) of soil(s) surrounding the well. Clays will exhibit the smallest ROI, while sands will exhibit the largest. An extraction well's ROI can typically range from 20 to 100 feet or greater.

Required Equipment

The required equipment is divided into two distinct areas (the pump and treat and the VE equipment), although some overlapping does occur. Figure 1 shows a typical simple modified groundwater pump and treat system with VE.

The equipment for a simple pump and treat system typically consists of electrical or pneumatic submersible pumps, depending upon the product and volume to be pumped. Pneumatic pumps have been used under explosion-proof conditions. A piping network directs the contaminated groundwater toward a treatment building. In the treatment building, the withdrawn groundwater is typically treated by granular activated carbon (GAC) units or by a countercurrent air stripper, with optional GAC polishing.

A simple VE system will consist of a vacuum pump capable of pulling approximately 200 actual cubic feet per minute (ACFM) of air per well at 5 to 15 inches of Hg. The piping network is connected to the extraction well(s). Other wells may be used as air inlet wells in order to monitor the ROI, create preferential air flow between two wells, or to inject air in order to enhance the efficiency of the VE system. After the air is withdrawn from the well, it is passed through a knockout pot, which removes water that may be entrained in the air stream.

After the knockout pot, the air stream passes through the vacuum pump. From here, the air may be discharged to the atmosphere or treated with vapor-phase GAC units or a thermal incinerator, depending upon the nature of the contamination and the requirements of the regulatory agency. This vapor-phase treatment may also incorporate the air stream from the air stripping tower. In addition, the liquid from the knockout pot may require treatment by the liquid-phase GAC units or the air stripping tower.

Advantages and Disadvantages

The advantages of the modified groundwater pump and treat system over a standard pump and treat system are the removal of VOCs (a potential future contamination source) from the existing vadose zone or cone of depression without excavating soil for off-site disposal. It also reduces or eliminates the need for groundwater recharge for soil flushing, while enhancing natural or added aerobic bioremediation in the soil.

Although two-phase VE technology is not discussed until later in this paper, the advantages of the modified pump and treat technology over the two-phase VE include its ability to:

- Remove groundwater more than 20 feet below the ground surface (BGS), including sites with multiple aquifers

- Remove free-product dense non-aqueous phase liquids (DNAPLs) or floating hydrocarbons
- Utilize VE only in those areas that contain a limited areal extent of vadose zone contamination
- Remove groundwater at significant pumping rates in permeable soils
- Discontinue the remediation of the vadose zone if the groundwater requires a longer to remediate, thereby saving operational costs

Disadvantages that this technology has in comparison to other technologies include:

- Its inability to remediate metals and heavier organics, such as long-chained hydrocarbons
- The potential need for off-site disposal of GAC units or withdrawn groundwater
- The potential need of air, groundwater withdrawal, surface water, or municipal wastewater treatment plant permits or approvals that could delay initiation of the remediation process

In summary, an ideal situation for the application of a modified pump and treat system with VE would include VOC vadose-zone contamination and with appropriate soil and groundwater conditions, such as permeable soils with deep groundwater contamination.

Costs, Reliability, and Limitations

A modified pump and treat system with VE is cost-effective compared to other technologies. Approximate costs for a system with one 40-foot recovery well, an air stripping tower for a 50 gpm flow, one 20-foot VE extraction well, and GAC polishing for the liquid and vapor phases is presented in Table 1. These costs can vary depending upon site obstructions, pump and treatment building requirements, and other factors.

The systems have been reliable in the field, but require typical maintenance associated with the pump and treat or VE systems. The primary perceived limitations is VE's inability to remove compounds with low vapor pressures. This limitation could hinder the remediation of a complex material, such as a heavy fuel, in which all components are not suitable for VE remediation.

Case Histories

Manufacturing Facility in Northern United States - Dames & Moore initiated a groundwater remediation program at a manufacturing facility in the northern United States in 1989. Although no VOCs were detected in the soil samples from the site and soil remediation was not required by the regulatory agency, Dames & Moore proposed a VE pilot study to determine whether VE could recover VOCs from source areas.

The VE pilot study was initiated in August 1989, and operated for a period of one month. During this period, the VE system recovered trichloroethylene (TCE) at an average rate of approximately 0.25 pounds per hour. The system remained at the site and has operated on various extraction wells across the site. As of February 1991, the system has removed over 2,500 pounds of TCE from the soil at the site. Cumulative TCE removed is shown on Figure 2.

The size of the original contaminant plume was estimated to contain approximately 300 gallons of TCE. The VE system has removed almost 200 gallons of TCE along, while the groundwater system has removed less than 40 gallons. Therefore, the use of the groundwater pump and treat system in conjunction with the VE has results in the substantial increased removal of the contaminant plume and reduced the anticipated duration of the groundwater remediation.

Manufacturing Facility in Northeastern United States - The following description pertains to a site that has a modified pump and treat system with VE. Although permitting delays by the regulatory agency have prevented the start-up of remediation program, this site is discussed due to complex environmental concerns that demonstrate thorough engineering considerations required during design of a remedial program.

At this particular site, VOC contamination was detected at concentrations greater than 10 mg/l in two aquifers (the overburden and bedrock aquifers). In addition, a layer several feet thick of free-product DNAPLs was detected in the shallow overburden aquifer on top of a dense confining silt lens. Elevated concentrations of natural iron and detectable concentrations of PCBs were detected in the some of the shallow groundwater. A geologic cross-section of the facility is shown on Figure 3.

The design of the remedial system had to consider and incorporate several factors. The final design used a modified pump and treat system with VE for removal and recovery of dissolved and free-product DNAPLs from the groundwater and vadose zone. The system was designed to control the existing contamination on-site using a groundwater recovery system in all three aquifer systems (the shallow overburden, confined overburden, and bedrock). The discussion of the remediation system will be divided into the recovery well system and the treatment equipment.

Recovery Well System - Recovery wells were installed in the three aquifer systems. In the shallow overburden, wells were placed in known areas of contamination, with closer well spacing in the free-product area. Additional recovery wells were placed downgradient of the free-product area and along the property boundary.

The wells were constructed within flush-mount manholes. Interconnecting piping was run through a large conduit. This design minimized interference to future users of the site and allowed for easier long-term maintenance of the system. The conduit contained numerous piping runs, including contaminated and recharge water lines.

Treatment Equipment - The system was designed to be contained in a small portion of the facility (approximately 55 feet by 70 feet). The piping conduits entered the building through a subsurface utility vault, from which the individual runs were piped to the various treatment equipment. A simplified diagram of the treatment system is shown on Figure 4. Briefly, the system includes:

- A vacuum pump to extract VOCs from the soil in the existing and newly formed vadose zones in the immediate area of the shallow recovery wells
- A liquid/liquid separator for free-product removal
- Aeration tank, Lamella, and sludge handling equipment for the removal and dewatering of elevated iron and suspended solids
- A disposable GAC unit for the removal of the soluble PCBs
- Countercurrent air-stripping packed towers for the overburden and bedrock aquifers to removal of VOCs from the groundwater
- Liquid GAC units for polishing the ground water prior to discharge to the storm sewer
- Steam-regenerable vapor-phase GAC units to recover VOCs from the VE system, air stripping towers, and other process tanks without off-site processing
- Recharge trenches to assist flushing contaminants, minimize potential settlement from dewatering, and allow the flexibility of adding bioremediation in the future

Eleven pneumatic pumps were installed in the overburden wells, just above the bottom of the wells. This placement of the pump will assist in maximizing the groundwater drawdown at the site and removing free-product DNAPLs. Pitless adapters, as shown on Figure 5, were used on the wells to facilitate the connection of the vacuum line, contaminated groundwater discharge line, and the pressurized air line for the pumps. An electrical pump was used in the bedrock well.

The VE system is connected to the pumping wells and will remove VOCs remaining in the soil during groundwater drawdown. It is also anticipated that the lower air pressure in the silty sand will increase the withdrawal rate of the groundwater and free-product DNAPLs.

The treatment system contains numerous sensors and other instrumentation to monitor flow rates, air pressure, and other important system parameters. The information gathered is processed by computer for local and/or remote system monitoring to allow the system to operate relatively independently. This computer program will perform monitoring and process control tasks, thereby reducing the need for operator assistance, increasing overall efficiency, and lowering operational costs.

TWO-PHASE VACUUM EXTRACTION

Review of the Technology

The two-phase VE process allows for the simultaneous remediation of soils and groundwater contaminated with VOCs. It is similar to combining a modified pump and treat system with VE into one unit. The two-phase VE process was co-developed in 1989 by Dames & Moore and is patent pending. Two-phase VE is similar to standard VE in the equipment required, except it is designed to actively remove contaminated groundwater from the extraction well along with the vapor-phase contamination. A typical two-phase VE system is shown on Figure 6.

In order to remove the air and liquid phases effectively, a greater vacuum (typically 16 to 22 inches Hg) is required. It can extract groundwater from depths down to 20 feet BGS. In tight soils, such as clays, the two-phase VE system can actually increase the groundwater withdrawal rate from the soil, when compared to standard pumping methods.

At the high vacuum, air and water enter into the well. If the water vapor is typically within 20 feet of the surface, water enters the piping network for removal by the knockout pot. The water is transported to the knockout pot in droplets that are entrained in the vapor flow and in slugs of water, which can be removed from the well as a results of the high vacuum. Due to the vacuum conditions and the surface area created when the water is transported as droplets, a significant portion of the VOCs in the groundwater enter the vapor phase during the removal process. Under these conditions, the extraction well operates similar to an air stripping tower with up to 99 percent transfer of VOCs from the liquid to vapor phase.

This technology is typically used in tight soils, where only a limited amount of groundwater can be withdrawn by traditional groundwater pumping. Groundwater withdrawal rates on systems installed to date range up to approximately 15 gallons per minute. The ROI in the vadose zone can typically reach 50 feet in the vadose zone and 70 feet in the groundwater, depending upon the soil conditions.

This technology was awarded the Consulting Engineers Council 1990 Honor Award for engineering excellence in Pennsylvania.

Required Equipment

The trailer mounted two-phase VE system typically uses a vacuum pump rated at approximately 200 ACFM at 18 inches Hg. The knockout pot may be larger, due to the greater working volume required to accommodate the groundwater. Similarly, the discharge from the knockout pot may require a larger pump to handle the additional groundwater flow. The knockout pot discharge pump can typically discharge 15 gpm at 16 inches of Hg.

As previously discussed, the vapor phase can be treated by GAC units or thermal incineration, if required by the regulatory agencies.

Advantages and Disadvantages

In an area with VOC contamination in the soil and shallow groundwater, the two-phase VE system can offer numerous advantages over a modified pump and treat system. The advantages not previously discussed include the system's ability to:

- Simultaneously remediate VOC-contaminated soil and groundwater
- Potentially eliminate the need for an air stripping tower
- House all motors in one building or trailer, since no down hole pumps are required
- Increase the groundwater withdrawal rate and capture zone in tight soils, such as clays and silts
- Allow field adjustments to the system by increasing or decreasing the vacuum at select wells
- Allow rapid evaluation of the remediation progress by analysis of the air discharge from the VE pump
- Minimize the number of wells

The two-phase equipment can arrive at the site in a self-contained trailer that is simply connected to pre-installed wells and utilities.

Some disadvantages were discussed in the section on the advantages of the modified pump and treat system with VE. Disadvantages to other remedial technologies are similar to those stated for the modified pump and treat system.

Costs, Reliability, and Limitations

Costs for a given site depend on specific site conditions, such as the areal extent of contamination, building obstructions, and other factors. For a "typical" two-phase VE system for a site that involves no site obstructions, one extraction well, shallow groundwater, and GAC treatment of the liquid and vapor phases, the approximate costs are presented on Table 2.

The capital costs for the treatment system includes a 200 ACFM vacuum pump, two transfer pumps, a carbon steel knockout pot, and associated instrumentation and piping. Additional equipment may be required, such as an air stripper or oil/water separator, to decrease remediation costs or meet regulatory standards.

Operation and maintenance costs for the system include estimates for additional GAC units, liquid- and vapor-phase analysis, weekly monitoring, electric, and routine maintenance. These costs will vary depending upon the monitoring requirements, contaminant concentrations, and other variables.

The systems installed to date have proven to be relatively reliable, with the first system running since September 1989. Two-phase VE also aerates the groundwater less, thereby reducing the potential for biofouling. However, additional time is typically required to debug the system and make field adjustments. Problems encountered to date include system shut down due to heavy rains and increased groundwater withdrawal rates.

In addition, some systems have experienced problems with removing significantly more contamination than anticipated. Typically, the system will remove the greatest concentrations of contaminants during the early weeks of operation. Unless the liquid- and vapor-phase treatment system(s) are over designed, they run the potential of being overwhelmed with far more contamination than the design basis. This could be due to the ability of the system to quickly volatilize substantial quantities of organics in the vicinity of the extraction well, producing greater concentrations than baseline values.

Limitations also include the inability to remediate complex contaminants, such as heavy fuels that contain compounds with low vapor pressures. Freeze protection is also required in cold climates.

CASE HISTORIES

Manufacturing Facility in New York - The first application of the two-phase VE technology occurred at an upstate New York facility. The client had intended to remediate the groundwater using conventional pump and treat methodology. Early pump tests yielded very low flow rates (approximately 0.2 to 0.3 gpm) for the clayey soils at the site. Therefore, conventional remediation technologies would involve a lengthy process.

Since conventional VE was contemplated for remediation of the VOCs in the vadose zone, design modifications of the system were explored to address both the groundwater and soil. Operations of a pilot conventional VE operation was initiated in September 1988. Modifications of the system enabled the two-phase recovery of the VOCs.

During the pilot test period, the two-phase VE system achieved a greater aquifer drawdown and a higher groundwater flow rate than earlier pumping tests. The drawdown and flow rate results are shown on Table 3. Well locations are shown on Figure 7. Following the pilot test, the two-phase VE system was fully operational in January 1989. The system decreased the water table and expanded the vadose zone. The groundwater withdrawal rate increased one order of magnitude (approximately 4 gpm compared to 0.3 gpm), and the major portion of the contaminant plume was captured. Figure 7 shows the measured area of groundwater capture of the extraction well.

During a 19 month of operation, 1,100 pounds of VOCs were removed from the soil and groundwater. Figure 8 shows cumulative VOCs removed from the vapor and liquid phases. Analytical results indicated that the system stripped 98.7 percent of the VOCs from the groundwater before the groundwater reached the surface, thereby eliminating the need for a separate air stripping tower.

The full scale design of the system required only three extraction wells located outside the building. The system originally contemplated would have required 47 wells, most of which would have had to be installed through the floor of the structure.

Former Manufacturing Facility - Trichloroethylene and tetrachloroethylene had been detected in soil and groundwater samples collected at a former manufacturing facility in the northeastern United States. The soil was classified as a silty clay and groundwater occurred approximately 10 to 15 feet BGS. Dames & Moore conducted a two-phase vacuum extraction pilot study at the site in December 1990.

The pilot test used a single extraction well. At an extraction well vacuum of 16 inches Hg, an average groundwater flow rate of approximately 7.5 gpm was withdrawn from the well. At groundwater withdrawal rates of 6 to 8 gpm, water table drawdowns were measured at three observation wells. Drawdowns of 1.41 feet, 1.15 feet, and 0.96 feet were observed at distances of 70, 100 and 140 feet, respectively.

Vacuum readings were measured at eight observation wells to determine the ROI in the vadose zone. The vacuum readings indicated that the two-phase VE system had a ROI between 30 and 70 feet above the water table.

The results of this pilot study were presented in a feasibility study. The two-phase VE system has been recommended for the full-scale remediation at the site. The proposed design will require three vacuum pumps, five extraction wells screened in the saturated zone, and ten extraction wells screened in the vadose zone.

Active Petroleum Bulk Storage Terminal - Dames & Moore is completing a pilot study at a bulk storage terminal in the southeastern United States. A spill of diesel fuels and gasoline had contaminated the vadose zone and groundwater.

Two extraction wells were extended 40 feet into the sandy soil and connected to the trailer mounted, two-phase VE system by 500 feet of PVC pipe. During the pilot study, some problems were experienced with the catalytic incinerator for vapor-phase treatment, due to equipment malfunction. In addition, the knockout pot discharge pump was unable to pump the groundwater withdrawn from the extraction wells under maximum vacuum of the pump.

During the pilot study, the vacuum was maintained between 12 and 14.5 inches Hg. The vacuum at the wells ranged between 9.5 and 11 inches Hg, resulting in a groundwater removal rate averaging 4 to 4.5 gpm.

The ROI in the vadose zone were approximately 50 feet, which is greater than expected, due to the significant short-circuiting that occurred through the uncovered sandy soil. The ROI in the saturated zone ranged between 100 feet laterally and 120 feet downgradient. The two-phase VE system appears to be maintaining control of the contaminant plume and reducing VOC concentrations. However, all of the analytical data has not been received and evaluated at this time.

Dames & Moore anticipates recommending that a full-scale treatment system be installed, if the data confirms the initial results. The full-scale system would probably include an air stripper for the liquid-phase prior to GAC treatment, in order to reduce operational costs.

FUTURE DIRECTIONS

The future for both technologies is bright, especially for two-phase vacuum extraction. Dames & Moore is evaluating the use of this technology in new applications for difficult-to-remediate situations. One new direction we are exploring is the installation of a system by horizontal drilling to address VOC contamination under active manufacturing building or aircraft hangers. These situations do not lend themselves to conventional treatment technologies, due to the inaccessibility to interior areas.

Figure 9 shows a conceptual horizontal drilling system. The anticipated effective zone of influence is shown on Figure 10. Preliminary cost estimates for one facility demonstrated a significant potential cost savings over a modified pump and treat system with VE, while not interfering with the on-going manufacturing process in the building.

CONCLUSIONS

The use of VE in conjunction with groundwater pump and treat systems is becoming more common throughout the country. Properly applied to a site with VOC contaminants in the vadose zone, it has proven to be an excellent remedial alternative.

The two-phase vacuum extraction system has a more limited track record and is suitable at fewer sites. However, at the appropriate site, it has demonstrated itself to be the most cost-effective method, while offering advantages not available with other technologies.

As more and more sites are successfully remediated with these technologies, modified pump and treat with VE and the two-phase VE systems, they will become a more staple method in the remediation arsenal.

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BIOGRAPHICAL SKETCH

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TABLE 1

**COST ESTIMATE FOR A GROUND WATER PUMP AND TREAT SYSTEM
WITH VACUUM EXTRACTION**

CAPITAL COSTS	
Well Installation and Trenching	\$12,500
Treatment System Capital (Vacuum Pump, Air Stripper, Knockout Pot, Pumps, Piping, etc.)	\$30,000-40,000
Treatment System Installation (Labor)	\$15,000
Trailer or Building	\$5,000-35,000
Utility Hook Ups	\$1,500
OPERATIONAL & MAINTENANCE (ANNUAL)	
Weekly Monitoring (4 hours x \$50)	\$10,000
Data Analysis (4 hours x \$50)	\$10,000
Reporting 12 Monthly and 2 Bi-annual	\$22,000
Water Analysis ¹ (3 samples x 12 mo. x \$250 ea)	\$9,000
Vapor Analysis ² (12 mo. x \$2,000/mo.)	\$24,000
Granular Activated Carbon	
Vapor ³	\$90,000
Liquid ⁴	\$6,000
Disposal	\$15,000
Routine Maintenance & Electric	\$10,000

Notes:

1. Assumes 3 samples every month analyzed for VOCs by EPA 624.
2. Assumes vapor phase concentrations maintained by portable gas chromatograph.
3. Assumes 0.5 lb/hr annual average (generally much less after first year) of VOCs in vapor phase. Carbon loading factor is 15%.
4. Assumes liquid-phase VOC concentration of 2 ppm. Carbon loading factor is 5%.
5. Costs depend upon site and contamination conditions, regulatory requirements, etc.

COST ESTIMATE FOR A TWO-PHASE VACUUM EXTRACTION SYSTEM

CAPITAL COSTS	
Well Installation and Trenching	\$10,000
Treatment System Capital (Vacuum Pump, Knockout Pot, Pumps, Piping, etc.)	\$20,000
Treatment System Installation (Labor)	\$15,000
Trailer (if necessary)	\$5,000
Utility Hook Ups	\$1,500
OPERATIONAL & MAINTENANCE (ANNUAL)	
Weekly Monitoring (4 hours x \$50)	\$10,000
Data Analysis (4 hours x \$50)	\$10,000
Reporting 12 Monthly and 2 Bi-annual	\$22,000
Water Analysis ¹ (3 samples x 12 mo. x \$250 ea)	\$9,000
Vapor Analysis ² (12 mo. x \$2,000/mo.)	\$24,000
Granular Activated Carbon	
Vapor ³	\$90,000
Liquid ⁴	\$6,000
Disposal	\$15,000
Routine Maintenance & Electric	\$10,000

Notes:

1. Assumes 3 samples every month analyzed for VOCs by EPA 624.
2. Assumes vapor phase concentrations maintained by portable gas chromatograph.
3. Assumes 0.5 lb/hr annual average (generally much less after first year) of VOCs in vapor phase. Carbon loading factor is 15%.
4. Assumes liquid-phase VOC concentration of 2 ppm. Carbon loading factor is 5%.
5. Costs depend upon site and contamination conditions, regulatory requirements, etc.

TABLE 3

PUMP TEST COMPARISON FOR A TWO-PHASE VACUUM EXTRACTION PILOT STUDY
CONDUCTED AT A MANUFACTURING FACILITY IN UPSTATE NEW YORK

	Duration (Days)	Average Discharge (gpm)	Drawdown (Feet Below Static)				
			RW-2	RW-3	RW-5	RW-7	RW-9
Traditional Pump Test on RW-10	2.1	0.3	0.04	0.08	0.05	0.08	0.07
Two-Phase VE System Test on VE 1A	3.0	4.1	0.39	0.48	0.20	0.30	0.58

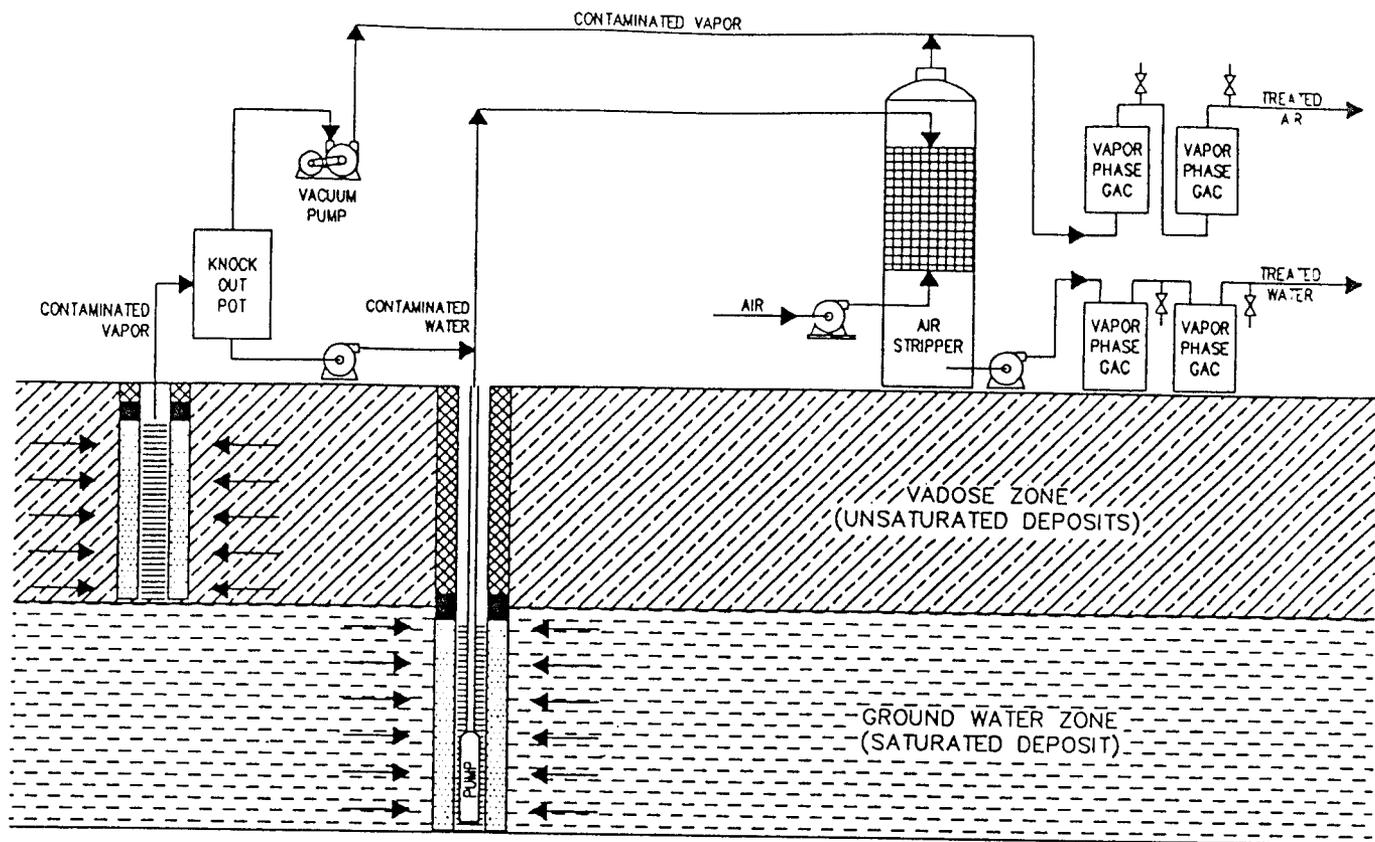


FIGURE 1: TYPICAL GROUNDWATER PUMP AND TREAT SYSTEM WITH VACUUM EXTRACTION

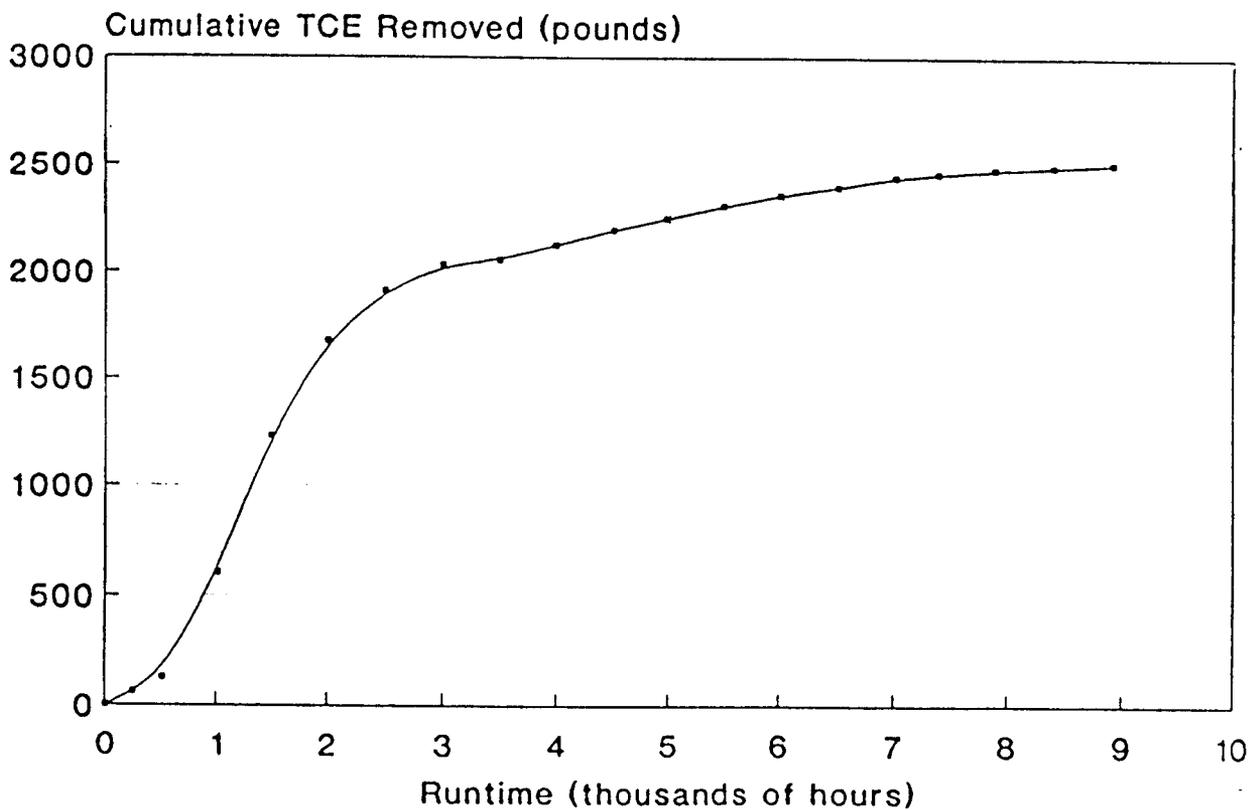
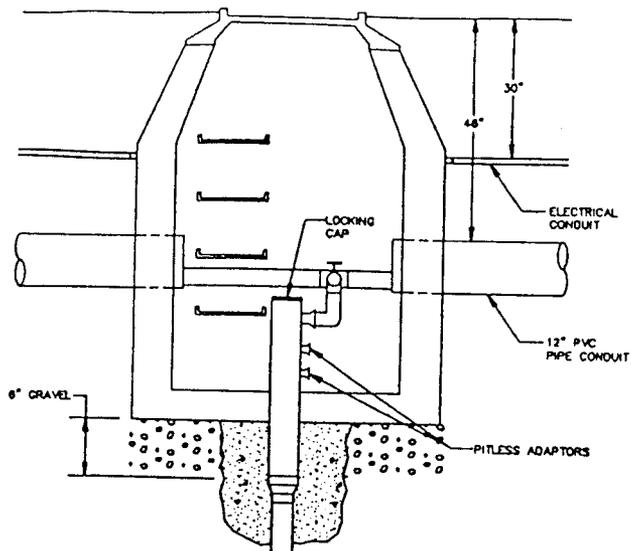
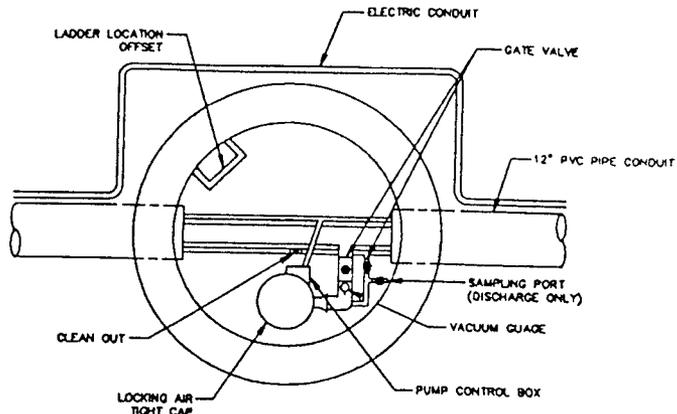


FIGURE 2: CUMULATIVE TCE REMOVED BY VE SYSTEM OPERATING IN NORTHERN U.S.



TYPICAL PUMP MANHOLE
ELEVATION



TYPICAL PUMP MANHOLE
PLAN

FIGURE 5: CONSTRUCTION DETAILS OF REMEDIAL SYSTEM

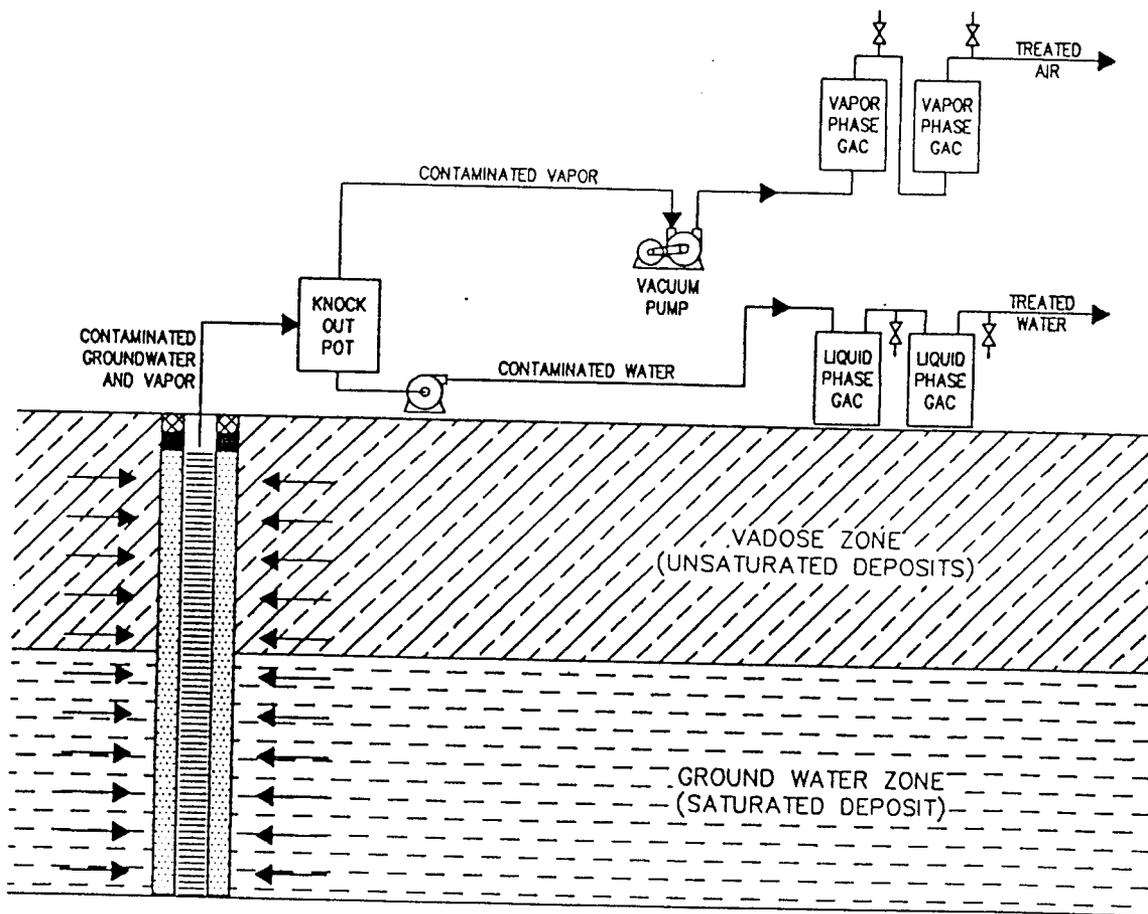


FIGURE 6: TYPICAL TWO-PHASE VACUUM EXTRACTION SYSTEM

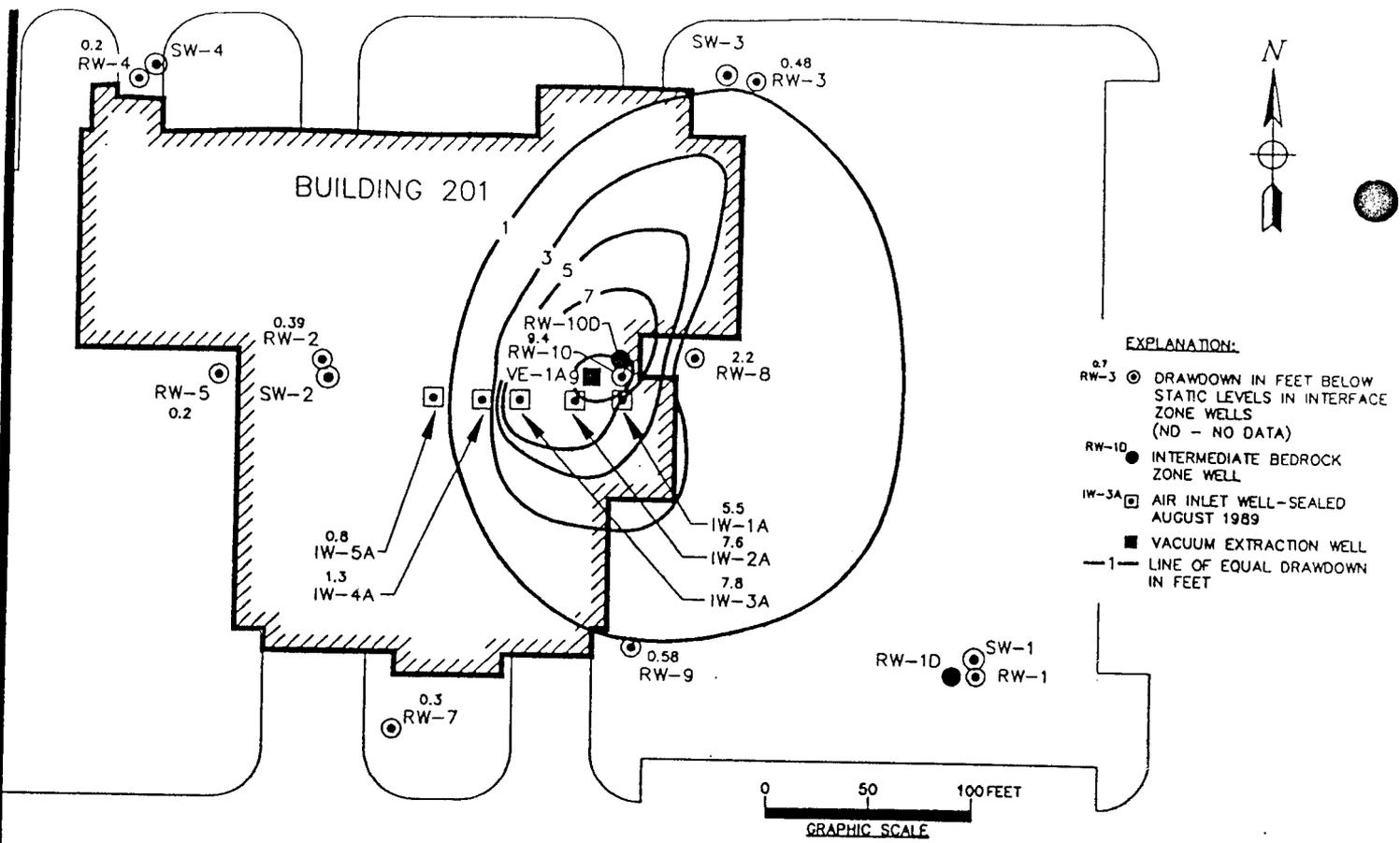


FIGURE 7: DRAWDOWN IN THE OVERBURDEN AQUIFER AT NEW YORK PILOT STUDY

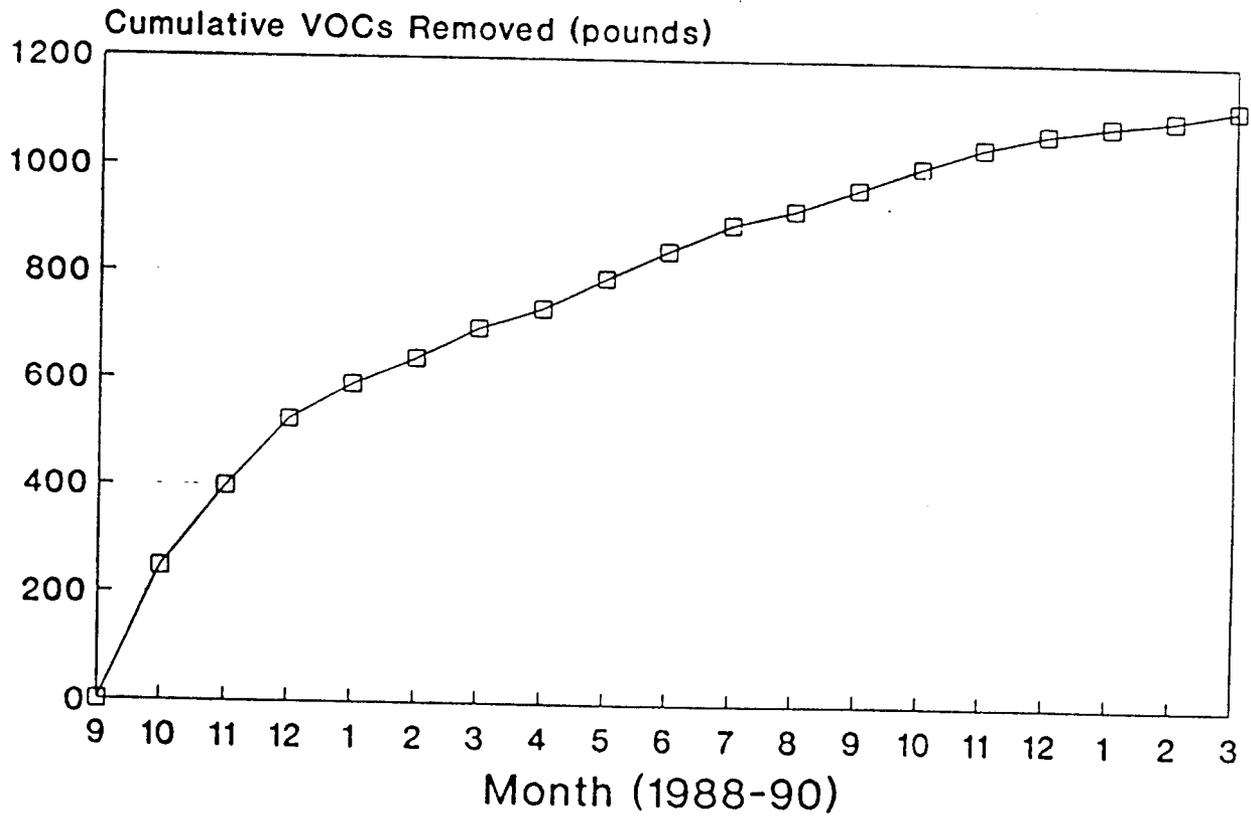


FIGURE 8: CUMULATIVE VOCs REMOVED BY TWO-PHASE VE SYSTEM IN NEW YORK PILOT STUDY

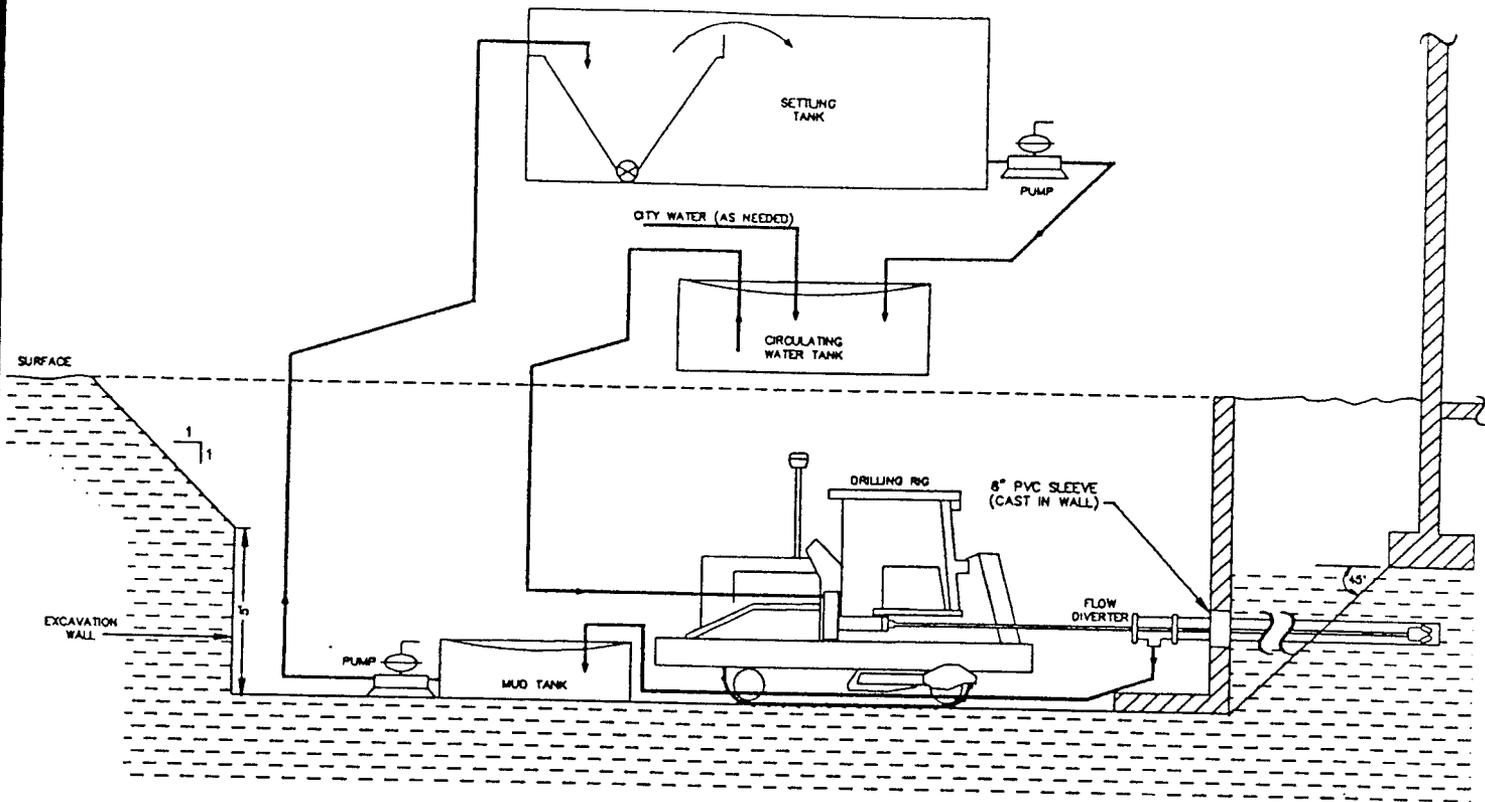


FIGURE 9: CONCEPTUAL HORIZONTAL DRILLING SYSTEM

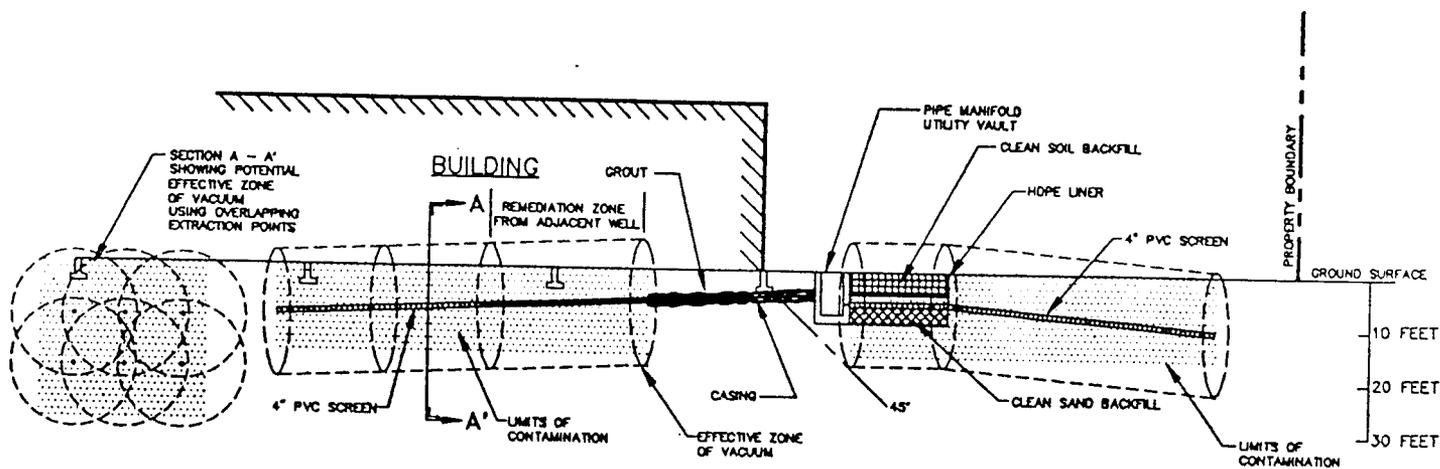


FIGURE 10: CONCEPTUAL HORIZONTAL DRILLING ZONE OF INFLUENCE