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NORTH PENN AREA 1  
PHASE II REMEDIAL  
INVESTIGATION REPORT  
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**Section 1**  
**Introduction**

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## Section 1 Introduction

### 1.1 Purpose of Report

This report presents the results of the North Penn Area 1 Phase II Remedial Investigation (RI) conducted between March and October 1992. The site was investigated by the Environmental Protection Agency (EPA) on the basis of a 1989 listing of the site on the National Priorities List (NPL). The site is one of 12 sites identified in the North Penn area by EPA in 1986 (NUS, 1986) on the basis of contamination of groundwater in production wells by volatile organic compounds (VOCs). The contamination at the Area 1 site was first noted in North Penn Water Authority (NPWA) well S-9. The well was decommissioned in 1979 because of high tetrachloroethene (PCE) levels in the groundwater. Subsequent potentially responsible party (PRP) searches by EPA identified five facilities in the area that may have contributed to the groundwater contamination. These facilities and the groundwater contamination were investigated in this RI.

The North Penn Area 1 site ("the site") is located in Souderton, Montgomery County, Pennsylvania. The site consists of the five facilities identified as potential source areas and that part of the bedrock aquifer containing VOC contamination underlying and in the vicinity of those facilities. The boundaries of the site are shown in Figure 1-1.

Investigations performed included soil boring, soil sampling and analysis, aquifer testing, and groundwater sampling and analysis. The soil sampling defined soil characteristics and levels of soil contamination by VOCs at the facilities. The aquifer testing characterized aquifer hydraulics and provided information needed to evaluate remedial options. The groundwater sampling defined the nature and extent of the VOC contamination in the bedrock aquifer at the site to the extent possible using existing wells; no monitoring wells were installed during the investigations.

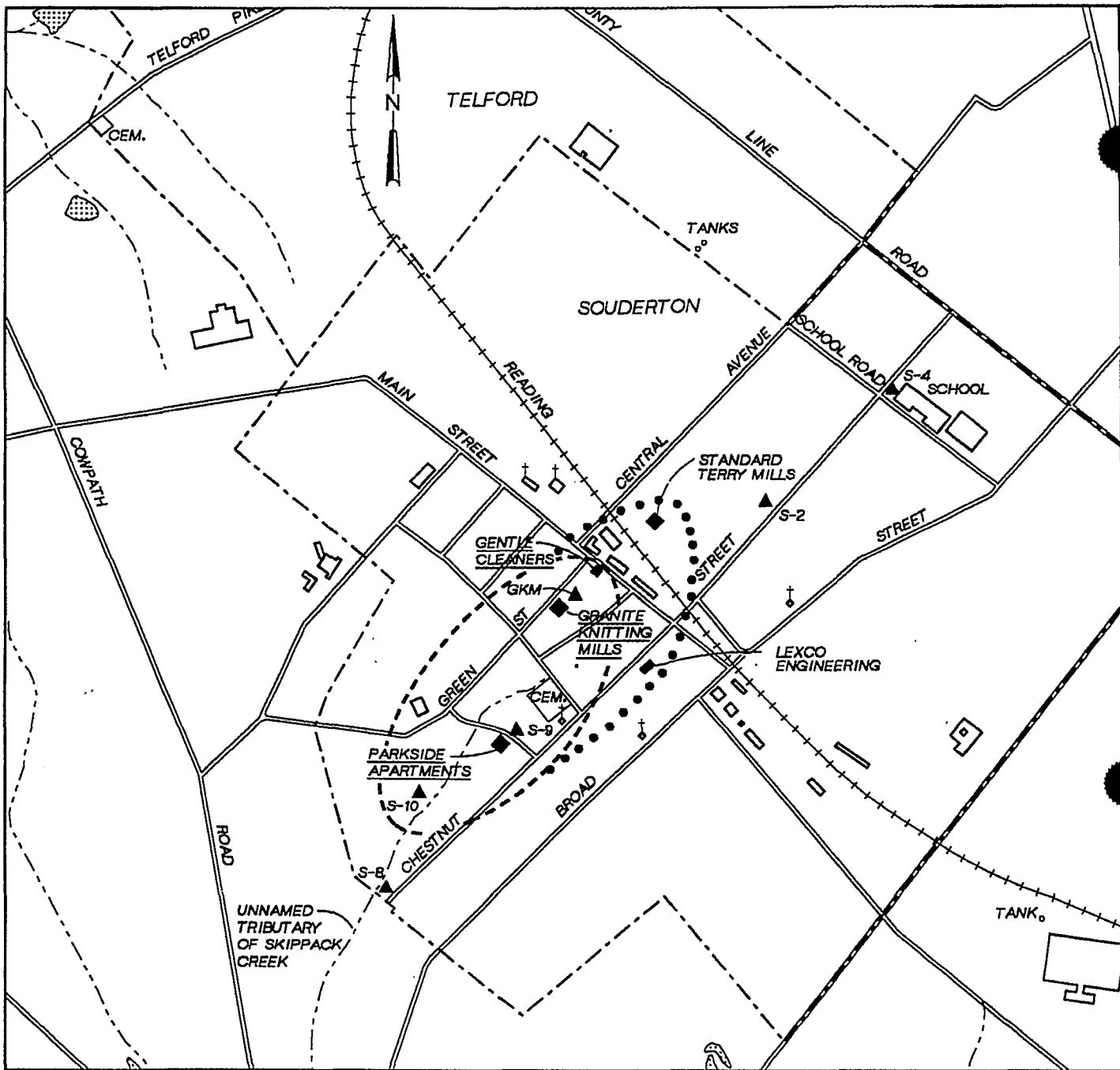
A chronology of site activities is presented in Table 1-1.

### 1.2 Site Background

#### 1.2.1 Site Description

Five facilities that may have contributed contamination to the groundwater were identified at the site: Gentle Cleaners, Granite Knitting Mills, Parkside Apartments, Lexco Engineering and Manufacturing Corp., and the former Standard Terry Mills. Locations of the facilities are shown in Figure 1-1. Figure 1-1 also shows the estimated limits of the site based on the locations of the facilities and the approximate distribution of contaminated groundwater in the bedrock aquifer.

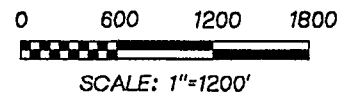




SOURCE: U. S. GEOLOGICAL SURVEY 7-1/2 MINUTE QUADRANGLE MAP FOR TELFORD, PA.

**LEGEND**

- ▲ S-2 NPWA WELL
- ▲ GKM GRANITE KNITTING MILLS WELL
- BOROUGH AND TOWNSHIP BOUNDARIES
- - - PRELIMINARY BOUNDARY OF AREA 1 (NUS, 1986a)
- REVISED BOUNDARY OF AREA 1 (1991)
- POTENTIALLY RESPONSIBLE PARTY (PRP)



**FIGURE 1-1**  
**SITE MAP**  
 North Penn Area 1  
 Phase II RI/FS



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**Table 1-1  
SITE CHRONOLOGY**

1979	The NPWA discovers PCE contamination in well S-9; sampling of wells S-9 and S-10 is initiated
June 1986	EPA Region III requests information from PRPs under CERCLA Section 104(e)
	104(e) information is provided by Parkside Apartments
July 1986	104(e) information is provided by Granite Knitting Mills, Inc., and by Gentle Cleaners, Inc.
	NUS Corporation completes the <i>Site Discovery</i>
August 1986	An EPA contractor samples residential and other wells at the site
September 1986	The site is scored using the Hazard Ranking System
January 1987	The site is proposed for the NPL
October 1987	Techlaw completes the <i>Final Facility Report</i>
June 1988	ATSDR completes a <i>Preliminary Health Assessment</i>
August 1988	The <i>Versar Technical Evaluation Report</i> is completed
March 1989	The North Penn Area 1 NPL listing becomes final
April to June 1992	CH2M HILL performs RI activities at site

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Gentle Cleaners began operating before 1953 (Versar, 1988; Techlaw, 1987). It is known that between 1953 and 1983, the company used 70 to 100 gallons of PCE per month as well as less than 1 gallon per month of chemicals containing 1,1,1-trichloroethane (1,1,1-TCA) and other chlorinated solvents of unknown composition. Since 1983, the volume of PCE used has been reduced to about 50 gallons per month. The PCE was stored onsite in either an aboveground storage tank or drums. An underground storage tank (UST) also may have been used to store PCE at the facility.

At Gentle Cleaners, Musheno (1980) documented a spill of 75 gallons of PCE occurring in the early 1970s. PCE reportedly flowed out the rear door onto the grassed area behind the building. In addition, discharge of PCE to a sink that drained into the same grassed area may have contributed to soil contamination.

Granite Knitting Mills (GKM) has operated a knitting mill since the early 1960s. From 1967 to 1979, a dry cleaning machine using PCE was maintained at the facility. Use of the machine may have stopped by 1979. PCE for the machine was stored in a tank inside the building. Wastes generated from the machine were estimated to contain about 2 percent PCE and were stored inside in drums on the southwest side of the building (Versar, 1988).

Property owners in the area report past discharges from the facility into the alley that runs along the southeast side of the building. These discharges were variously described as solvents and dyes, but their point of origin along the building was not identified. Reportedly, drums containing waste oil with some solvent contamination were stored outside along the southwest side of the building prior to disposal.

The Parkside Apartments property once included a dry cleaning establishment. Before that, the property was used as a beer distributor, and before that, as a slaughterhouse (CH2M HILL, 1989b). Three USTs containing petroleum hydrocarbon fuels were once located on the property, but were allegedly removed around 1980 (personal communication, P. Stoudt, 1989). Another UST that may have been present at the south corner of the facility (Musheno, 1980) could not be located during onsite activities. Area residents reported that part of the facility may have been landfilled with dirt and construction debris.

Lexco Engineering and Manufacturing Corp. (Lexco) has used 1,1,1-TCA at its facility since 1979. The facility purchases 110 gallons annually and generates 10 gallons as waste annually. The 1,1,1-TCA is used in a trough and may have spilled or leaked onto the floor. Operations at the facility have been ongoing since before 1960 (Logan Deposition, 1991). The facility has one UST in use for oil storage. A second UST for gasoline storage was located along the side of the building and is no longer in service.

The former Standard Terry Mills building was previously occupied by a trolley repair shop, supermarket, gas station, knitting mill, and other unidentified activities. These operations are believed to have been potential users of solvents and chemicals that may have contributed to local groundwater contamination.

The former Standard Terry Mills facility was in operation until May 1991, when a fire occurred at the facility; the buildings were later razed. The facility has undergone a Phase I property assessment, which primarily evaluated the site for polychlorinated biphenyl contamination; the report indicated that the facility was uncontaminated (personal communication, P. McManus, 1992). Two USTs were identified at the facility. In addition, John Crawford, current owner of the former Standard Terry Mills, reports that one UST was removed about 7 years ago and that two fuel oil tanks were removed around January 1991 (EPA, July 1991).

### **1.2.2 Summary of Past Remedial Actions**

No remedial actions have been undertaken at North Penn Area 1. Water from the NPWA wells S-8 and S-10 is not treated but is blended with water from other wells to meet drinking water quality standards. Well S-9 is shut down.

### **1.2.3 Summary of Previous Investigations**

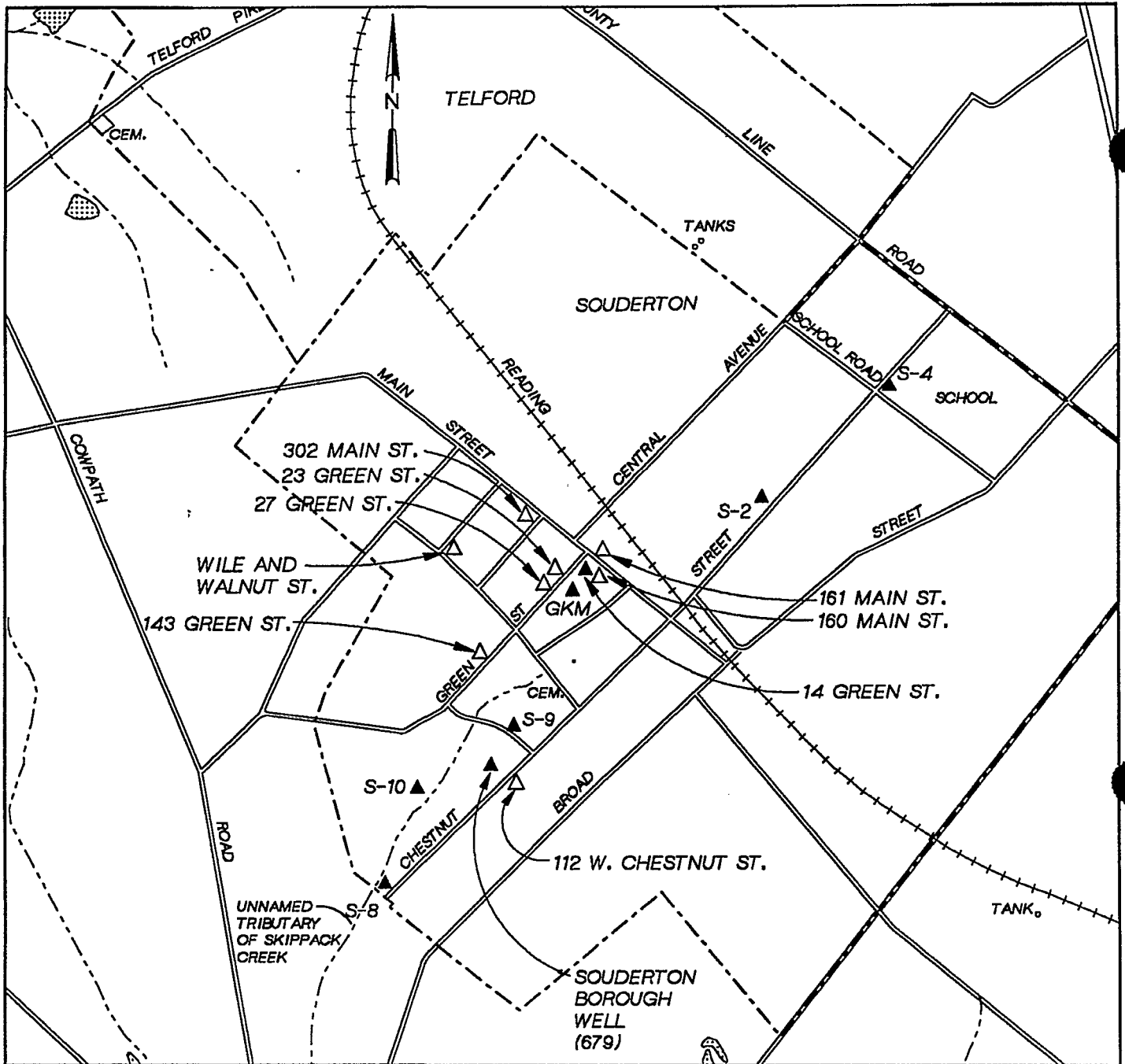
This section discusses past groundwater sampling activities and analytical results. The locations of wells in Area 1 for which analytical data were obtained in previous investigations are shown in Figure 1-2. The well designated "GKM" is located at Granite Knitting Mills. Analytical data for the wells shown in Figure 1-2 are included in Appendix A. The quality of these data could not be determined. Samples taken on August 28, 1986, were collected by NUS for EPA and were analyzed by the NPWA (NUS, 1986b). Other samples from the NPWA wells were both collected and analyzed by the NPWA.

Groundwater was analyzed for the following VOCs:

- 1,1,1-TCA
- 1,1-Dichloroethane (1,1-DCA)
- 1,1-Dichloroethene (1,1-DCE)
- cis- and trans-1,2-DCE
- PCE
- TCE

The groundwater analyses (see Appendix A) showed the following:

- Well S-9 contained 1,1,1-TCA, TCE, and cis-1,2-DCE at concentrations up to approximately 1 ug/l, and PCE at 10 to 25 ug/l.

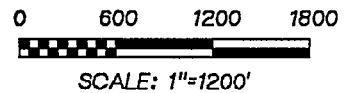


SOURCE: U. S. GEOLOGICAL SURVEY 7-1/2 MINUTE QUADRANGLE MAP FOR TELFORD, PA.

**LEGEND**

- ▲△ WELL LOCATION SYMBOL:
- R = RESIDENTIAL WELLS
- GKM = GRANITE KNITTING MILLS WELL
- S-X = NPWA WELLS

NOTE: OPEN SYMBOL INDICATES NO CONTAMINATION DETECTED.



**FIGURE 1-2**  
**WELL LOCATION MAP**  
**OF WELLS SAMPLED**  
**BEFORE THE RI**  
 North Penn Area 1  
 Phase II RI/FS



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- An unused Souderton Borough well (well 679 in Figure 1-2) contained PCE up to 4.3 ug/l and TCE up to 5.0 ug/l.
- Well S-10 contained 1,1,1-TCA at concentrations up to about 0.6 ug/l and PCE at concentrations up to approximately 30 ug/l.
- One residential well, used for gardening (a dug well at 14 Green Street in Figure 1-2), contained PCE at a concentration of about 14 ug/l.
- The Granite Knitting Mills well contained 1,1,1-TCA at approximately 250 ug/l; 1,1-DCA and 1,1-DCE at approximately 5 to 10 ug/l; and cis-1,2-DCE, trans-1,2-DCE, PCE, and TCE at approximately 10 to 35 ug/l.

Additional analytical results not provided in Appendix A include:

- NPWA well S-2 was uncontaminated during one sampling event in March 1988. PCE was detected in this well at 0.6 ug/l in routine NPWA sampling on February 14, 1991.
- NPWA well S-4 contained from 2.3 to 3.4 ug/l of PCE during sampling events between March and September 1988; no TCE was detected.

### 1.3 Report Organization

The RI report is divided into eight sections. Section 1 constitutes the introduction. A description of the investigations is in Section 2. The physical characteristics of the study area are described in Section 3. Section 4 discusses the nature and extent of contamination at the site. Contaminant fate and transport are explained in Section 5, followed by a risk assessment in Section 6. The findings of this report and the conclusions are presented in Section 7. Finally, the references are included in Section 8. The appendices contain analytical results and procedures that support the material presented in the RI report.

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**Section 2**  
**Study Area Investigation**

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## Section 2 Study Area Investigation

### 2.1 Surveying

As part of the RI, a surveying team measured the vertical locations of the tops of the casings of several wells at the site. Elevations of well casing measuring points were surveyed to the nearest 0.01 foot. Soil boring sites were located by measuring their distance from permanent facility features with a hand-held tape. No other surveying was conducted.

### 2.2 Contaminant Source Investigations

Contaminant source investigations at the site consisted of soil sampling at the five facilities identified as using or having used chemicals containing the contaminants detected in the groundwater. Contaminant migration through soils to the groundwater is believed to be the major source of groundwater contamination. Details of the soil-sampling investigation are discussed in Section 2.4.

### 2.3 Geology

The geology immediately underlying North Penn Area 1 was not directly investigated during this RI. Information was obtained on the drilling of NPWA well NP-69 and is discussed in Section 3 on the physical characteristics of the site.

Video and geophysical logging were performed on the Granite Knitting Mills (GKM) well and NPWA well S-9 in this investigation. The video logging consisted of a downhole television survey. Fracture locations and orientations in the wells were identified from the video logging. This logging was followed by packer testing in the GKM and S-9 wells.

The U.S. Geological Survey (USGS) collected geophysical log data on the GKM well and NPWA well S-9. Borehole logging by the USGS included caliper, single-point resistivity, natural gamma, temperature, and fluid resistivity probes. Velocity logging by brine tracing was performed by injecting small quantities of sodium chloride solution into the wells and measuring the rate of movement of the slug with the fluid resistivity probe under nonpumping conditions. The geophysical logging results received from the USGS are presented in Appendix B.

A survey of surface fracture traces was performed at the site. Because of the extensive urbanization at the site, aerial photographs could not be used to delineate fractures within Souderton. Aerial photographs were used to identify potential fractures within



2 miles of the site. Usable outcrops were located near the site and some fractures were identified using aerial photographs. Fractures in outcrops were measured for their orientation, width, and spacing. Data obtained were incorporated into a map and a rose diagram, which show the fracture distributions in the vicinity of the site.

## 2.4 Soil

The soil investigation, involving soil borings, soil sampling, geotechnical analysis, and chemical analysis by field gas chromatograph (GC) and by Contract Laboratory Program (CLP) laboratories, provided information on potential contaminant sources and on the need for soil remediation.

The objectives of this work were to:

- Determine the nature and extent of soil contamination
- Determine physical characteristics of the soil for evaluation of possible remedial alternatives
- Establish representative contaminant concentrations in soils for qualitative and quantitative risk assessment
- Identify underground and aboveground storage tanks that may have contained or might still contain chemicals of concern.

Soil samples were obtained during the RI using hollow-stem auger drilling and split-spoon sampling. When access difficulties restricted the use of a drill rig at the back of Gentle Cleaners, a tripod rig was used. Split-spoon samples were collected continuously from the ground surface to the top of bedrock, providing both surface and subsurface samples. Samples were visually described and the soil classified according to the Unified Soil Classification System.

Split-spoon samplers were used to collect soil samples for chemical and visual analysis according to ASTM D 1586-84. The soil samples were manipulated with decontaminated stainless steel utensils. The soils were field composited for total organic carbon (TOC) analyses. The specific sample handling procedures used at the site are outlined in the Sampling and Analysis Plan (SAP). The sampling equipment was decontaminated between samples according to the procedures outlined in the SAP. The decontamination procedures generally consisted of soap washing, a nitric acid rinse to remove metals, two solvent rinses to remove organics (methanol and hexane), and distilled water rinses.

The locations of the PRP facilities and areas of potential contamination at each facility were identified in the PRP search performed by EPA. These data were supplemented

by review of available aerial photographs and conversations with people familiar with activities at the site.

Information on the location and nature of alleged USTs at Parkside Apartments, Gentle Cleaners, Standard Terry Mills, and Lexco was collected and evaluated to assess the possibility that tanks containing chemicals of concern were or still are located at the site. Information on the possible locations of USTs was obtained from surface metal detector surveys performed as part of the RI and by conversations with local sources. The potential for these tanks having been sources of contaminants to the soils and to the groundwater was qualitatively addressed by screening soil samples and quantitatively addressed by analyses of soil samples for volatile organic compounds.

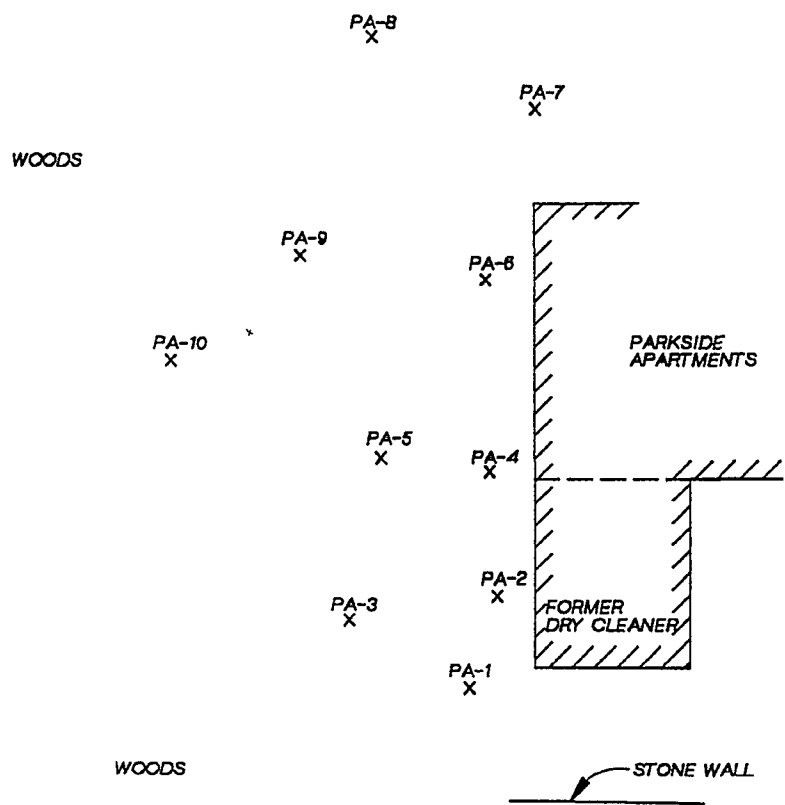
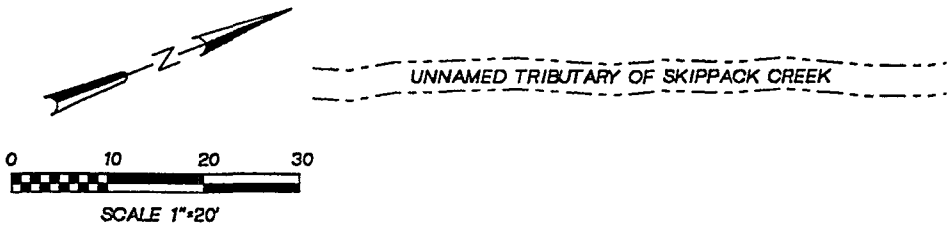
Locations of soil borings were based on information on waste-handling practices, aerial photographs, access to locations, and the results of the soil sampling. At Parkside Apartments (Figure 2-1), the soil borings were drilled along the back of the former dry cleaner building where discharge of waste solvents to the surface may have occurred. Soil contamination was encountered in the first few borings next to the buildings, and the area investigated using soil borings was expanded to better define the extent of the contamination.

At Granite Knitting Mills (Figure 2-2), soil borings were drilled around the solvent storage and handling facilities at the southwest side of the building. Soil borings were installed in the area along the side of the building where waste solvents were stored in drums. Additional borings were installed in the alley beside Granite Knitting Mills to evaluate the extent of contamination found in initial borings along the alley. The additional borings also served to evaluate the connection of soil contamination between Granite Knitting Mills and Gentle Cleaners.

At Gentle Cleaners, soil borings were drilled in the backyard, where the tripod rig could be set up (Figure 2-3). Allegedly, this is the area where waste solvents may have been disposed of out of the back door from a sink drain that discharges through the back wall and from a 75-gallon PCE spill that reportedly flowed out the back door. Additional soil borings were drilled in the yard south of Gentle Cleaners where probable stains were observed in aerial photographs. This area is a shallow depression that is probably frequently wet, and may have received surface runoff of waste solvents improperly handled at Gentle Cleaners. Eight soil borings were installed in the backyard and the side yard. Overhead power lines, construction debris, and trees limited the access to other boring locations at this facility.

The soil borings at Standard Terry Mills (Figure 2-4) were located to take samples in the areas of the underground fuel storage tanks and the vicinity of the former wooden loading dock. The borings were located to provide a scan of those areas.

The soil borings at Lexco (Figure 2-5) were located on the southeast side of the building in the loading area and in the vicinity of two USTs and a concrete pad. The borings were located to provide areal coverage of the parking area.

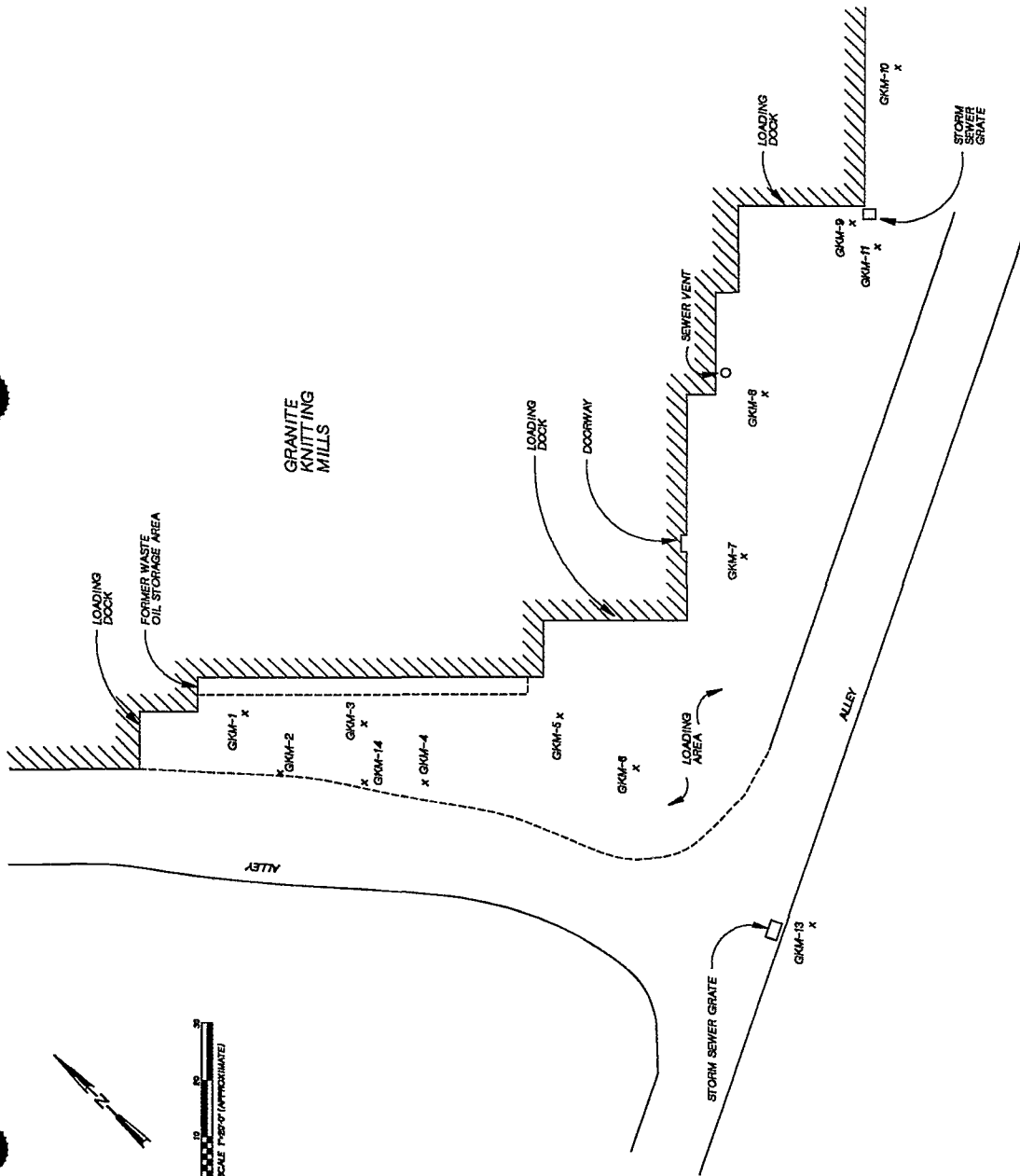


**LEGEND**  
X SOIL BORING LOCATIONS

**FIGURE 2-1**  
**PARKSIDE APARTMENTS**  
**BORING LOCATIONS**  
North Penn Area 1  
Phase II RI/FS

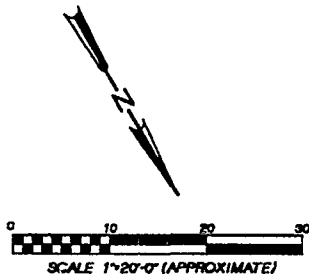


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**LEGEND**  
 SOIL BORING  
 LOCATIONS  
 x

**FIGURE 2-2**  
**GRANITE KNITTING MILLS**  
**BORING LOCATIONS**  
 North Penn Area 1  
 Phase II RI/FS



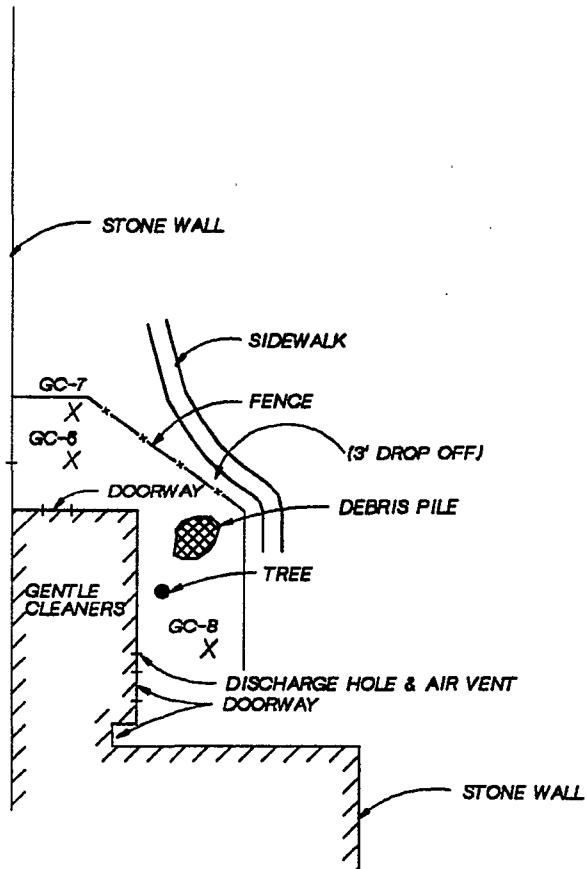
GC-1 X

GC-2 X

GC-3 X

GC-5 X

GC-4 X



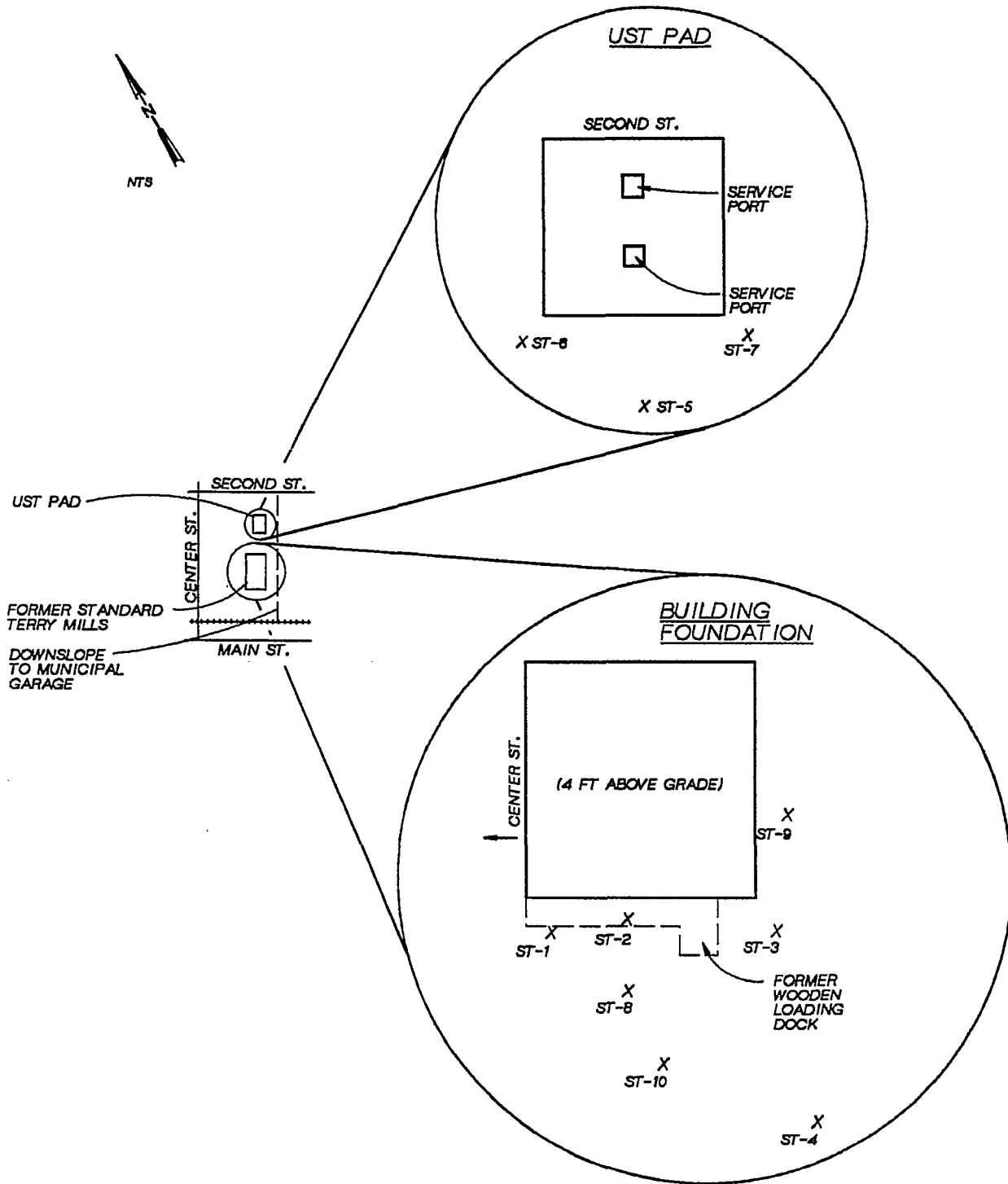
**LEGEND**

X SOIL BORING LOCATIONS

**FIGURE 2-3**  
**GENTLE CLEANERS**  
**BORING LOCATIONS**  
 North Penn Area 1  
 Phase II RI/FS



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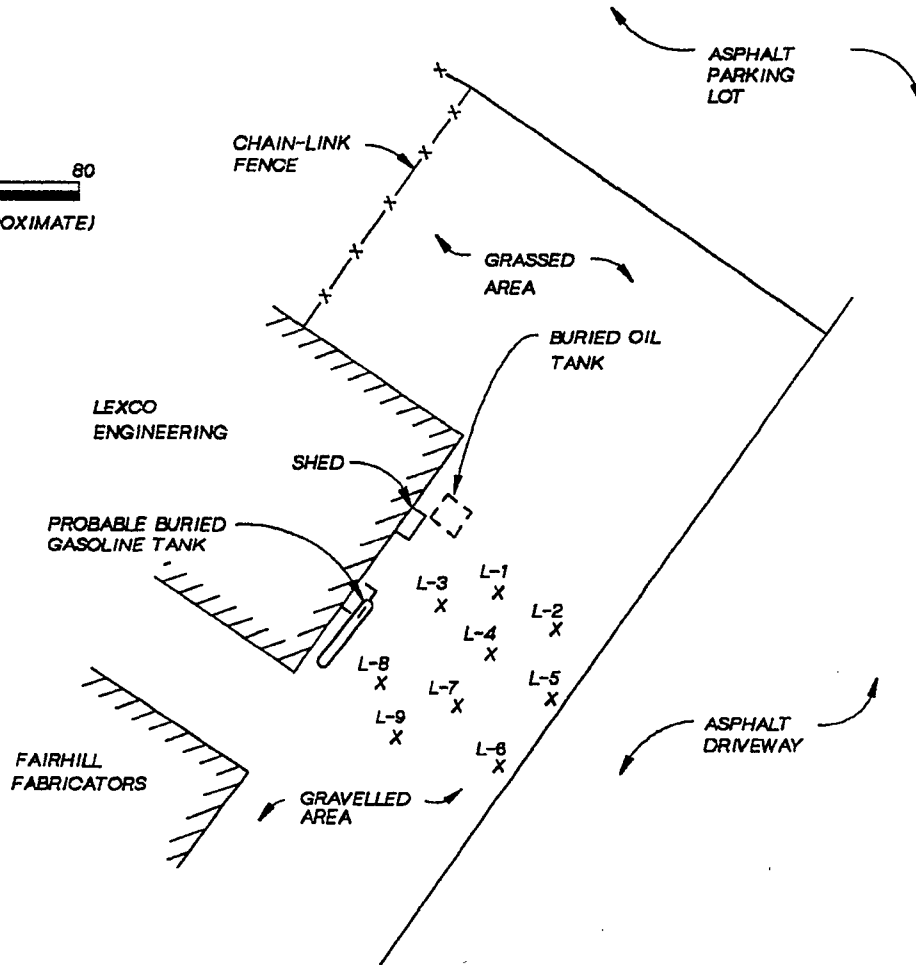
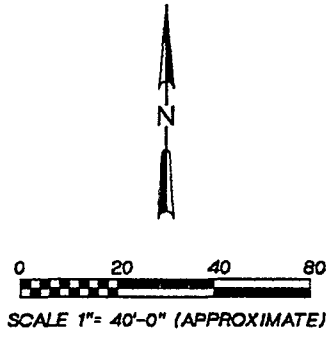
**LEGEND**

X SOIL BORING LOCATIONS

**FIGURE 2-4**  
**STANDARD TERRY**  
**BORING LOCATIONS**  
 North Penn Area 1  
 Phase II RI/FS



AR300693



**LEGEND**

X SOIL BORING LOCATIONS

**FIGURE 2-5**  
**SOIL BORING**  
**LOCATIONS AT LEXCO**  
**ENGINEERING**  
 North Penn Area 1  
 Phase II RI/FS



AR300694

On the basis of appearance, odor, and photoionization detector (PID) readings, approximately one-half of the samples were analyzed for selected VOCs by a portable field GC unit; the VOCs analyzed for were TCE, PCE, 1,1-DCE, cis- and trans-1,2-DCE, and 1,1,1-TCA. The field GC procedures are discussed in Appendix C. Some samples testing negative for appearance, odor, and PID readings also were analyzed. Samples were selected at discrete intervals. In some cases, TOC samples were composited when sufficient volume for analysis was not available in a single sampling event. When more volume was required for TOC sample analysis than was acquired through collecting a standard split-spoon sample, additional volume of soil was collected by compositing the TOC samples. Composite TOC samples were collected either from an adjacent borehole (less than 2 feet away) at the same depth as the original sample, or from the 2-foot interval below the original sample.

Depending on the results of the field GC analyses and sample distribution, samples were selected for analysis in the CLP or at the EPA Region III Central Regional Laboratory (CRL) for volatile organic compounds on the Target Compound List (TCL) (see Table 2-1) to determine the presence of other contaminants and to confirm the field GC results. Samples with positive and negative field analytical results were submitted. In addition, samples submitted to the CLP or CRL were analyzed for TOC to determine partitioning of the compounds in the soil and the potential for compounds leaching to the groundwater.

All soil materials brought to the surface and not retained as samples were returned to the boring in 2-foot lifts and compacted using the soil boring rig. Any open hole remaining was backfilled to the surface with grout, and the surface returned to its natural state.

Geotechnical samples were obtained to provide information in the event that soil remediation is needed. At three locations where field screening analyses indicated the presence of contamination, surface soil samples were obtained in 24-inch Shelby tubes and submitted to a geotechnical laboratory for physical characterization. The physical characterization included the following:

- Atterberg limits
- Particle size distribution
- Water content
- Specific gravity
- Porosity (total)
- Bulk density
- Permeability

During soil boring activities, the air quality in the vicinity of the drilling was monitored as part of implementing the Health and Safety Plan. This monitoring did not indicate that contamination was affecting air in the breathing zone. Volatile organics were noted in some of the soil borings. Headspace readings on some of the soil samples showed some volatile organics.



Table 2-1  
TARGET COMPOUND LIST

Volatile Organic Compounds

acetone  
benzene  
bromodichloromethane  
bromoform  
bromomethane  
carbon disulfide  
carbon tetrachloride  
chlorobenzene  
chloroethane  
chloroform  
chloromethane  
dibromochloromethane  
1,1-dichloroethane  
1,2-dichloroethane  
1,1-dichloroethene  
1,2-dichloroethene (total)  
trans-1,2-dichloroethene  
1,2-dichloropropane  
cis-1,3-dichloropropene  
trans-1,3-dichloropropene  
ethylbenzene  
2-hexanone  
methyl ethyl ketone (2-butanone)  
4-methyl-2-pentanone  
methyl chloride  
styrene  
1,1,2,2-tetrachloroethane  
tetrachloroethene  
1,1,1-trichloroethane  
1,1,2-trichloroethane  
trichloroethene  
toluene  
vinyl acetate  
vinyl chloride  
xylene (total)

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Information provided by Musheno (1980) and observations during a site visit (CH2M HILL, 1989b) suggested the presence of an UST near the south corner of Parkside Apartments. During the soil boring investigation at this facility, a metal detector was used to scan the area for a buried tank, but no tank was located. Two USTs were identified at Lexco; they were determined to contain gasoline and fuel oil (personal communication, CH2M HILL, 1992). Three USTs were reported at Standard Terry Mills. The Souderton Fire Marshall (personal communication, Paul Stoudt, April 20, 1992) identified two of the tanks. The Fire Marshall reported that he believed that the third tank was located in the same area; however, he was unable to identify tank contents or condition. No USTs were noted in the yard at the rear of Gentle Cleaners, where Versar (1988) suggested an UST may be located.

Site-specific information available on soils is of two types: a general determination of thickness and descriptions of the physical parameter. Because of the amount of construction activity around the site, soils are variable in thickness, ranging from 5 to over 15 feet. The soil that is present primarily consists of the residual soils described in the section on regional soils in Section 3. The depth of the soils was determined on the basis of the presence of residual rock fragments and on the blow counts of the sampling device.

## **2.5 Groundwater Investigations**

### **2.5.1 Well Inventory**

The well inventory defined wells at and in the vicinity of the site and identified locations for additional groundwater sampling. Information was collected on well construction specifications and exposure pathways related to groundwater. The task was to locate public and private wells (including residential, commercial, and industrial wells) to identify potential receptor locations and to identify potential vertical migration routes of contaminants within wells. Well locations were divided into current offsite residential use and potential future onsite use.

The Souderton area is urban and public water is available, but it is known from earlier studies that many of the town residents have private domestic wells despite the availability of public water. The domestic wells are generally hand-dug wells from before the time the public water supply was available. All of these wells are believed to be unused or no longer used for potable purposes.

To keep the well inventory to a manageable size, the radius of the survey was limited to 1/4 mile to the west, north, and east from Gentle Cleaners and Granite Knitting Mills. However, because it is probable that the contaminant plume is moving to the south and southwest beneath the valley of the unnamed tributary of Skippack Creek, the well inventory covered a distance up to 1/2 mile from these PRPs in these

directions. This distance includes possible wells owned and used by residents along the valley floor, which is heavily urbanized.

A door-to-door survey of residences, businesses, schools, and other industries was performed. The task also involved compiling and analyzing information from data bases maintained by the USGS, the Pennsylvania Geology Survey, Pennsylvania Department of Environmental Resources (PADER), and other organizations. Important information obtained included well depths, screen or open-hole intervals, pumping rates, and frequency of well operation. This information is provided in Section 3.6, and summarized in Table 3-2.

### **2.5.2 Well Logging**

Well logging performed in the NPWA well S-9 and the GKM well provided information on subsurface fracture distribution and potential vertical flow in these wells. The well logging consisted of downhole television and geophysical logging. This information was used to determine the intervals for straddle packing and to refine the conceptual hydrogeological model. A surface fracture study provided additional information on fracture distributions.

The video logging consisted of a downhole television survey. In a downhole television survey, a television camera is slowly lowered the length of the well, and the image provided by the camera is observed and taped for later review. Downhole television logging was performed as part of the RI in the GKM well and NPWA well S-9. The entire depth of each well was logged. This logging provided information on fracture distribution within the well; it also provided information to evaluate potential interim remedial actions for these two wells.

The downhole television logging was evaluated in the field to identify potential zones for straddle packer sampling. The television logging also was reviewed for observed fracture orientations. In addition, well conditions were observed from the television logging.

Single point electric, fluid resistivity, natural gamma, caliper, temperature, and brine trace logging were performed as part of the borehole geophysics. The geophysical logging required a downhole probe, a hoist and cable, and monitoring and recording equipment. The logs were collected by lowering the probe into the well at a controlled rate. To perform the different types of logging, different types of probes and monitoring and recording equipment were used. Analog plots of readings versus depth in well were produced.

For single point electric logging, a charged electrode was lowered through the well. A second electrode at the surface was used to measure the drop in current as the electrode moves down the borehole.

A fluid resistivity log also was evaluated. The fluid resistivity log was performed by lowering an uncharged electrode through the well and monitoring the difference in electric potential between the electrode in the well and an electrode at the surface. Fluid resistivity and single point electric logs were used to try to identify strata. Because the bedrock being logged in this RI is lithologically similar, strata identification was expected to be ambiguous.

Natural gamma logs were collected at each of the wells. Natural gamma logs were used to identify differences in naturally occurring radiation in the materials encountered in the borehole. Potassium 40, a naturally occurring radioactive isotope in clays, makes it possible to distinguish clay-bearing from nonclay-bearing beds (mudstone from sandstone). Gamma logs were used as a qualitative confirmation of stratigraphic correlation. The lack of information about stratigraphy in the wells logged in this RI and the similarities in the physical properties of the formations made evaluation of these logs difficult. Large borehole diameters can reduce the amount of radiation reaching the detector in the probe creating an abnormally low reading in that zone.

Caliper logging also was performed in each of the wells. In the caliper logging, a probe monitored borehole diameter. These data should correlate closely with the fractures observed in the downhole television logging. The caliper logging provided identification of zones where the borehole diameter changed significantly. Fracture zones were interpreted from caliper logs in locations where the borehole widened.

Temperature logging is the monitoring of fluid temperatures in the borehole. Temperature logs are commonly used to identify small aquifers or source zones within a larger aquifer. Slight variations in temperature in a discrete zone are used in the identification of these more productive zones.

### **2.5.3 Packer Testing**

Straddle packer testing in well S-9 and the GKM well provided information on the productivity of discrete vertical intervals and the means to obtain groundwater samples from those intervals. This information was used to refine the conceptual hydrogeological model of the site.

Most groundwater flow and contaminant movement at the site occur in distinct zones characterized by fractures and joints, rather than in the bedrock matrix. To define the hydrogeologic system, the hydraulic characteristics of the fractured zones were investigated. This investigation was accomplished by isolating fracture zones to separate the fractures from the hydrogeologic system acting as a whole. This isolation was accomplished by using straddle packers.

In the straddle packer approach, inflatable packers were lowered into each well to depths above and below the interval of interest. The interval of interest was defined from the television log as intervals of fractured rock separated by intervals of

unfractured rock. Where possible, the packers were inflated in intervals without fractures. If the interval of interest was at the top or the bottom of the hole, only one packer was installed below or above the interval, respectively. When the packer was inflated, the borehole interval between the packers was effectively sealed off and prevented from interacting with other water-bearing zones via the borehole. Interconnection between zones via the natural fracture and joint system was not affected.

Five intervals were tested in each borehole. Pumping was performed on each packed interval by installing a pump within the packed interval and pumping groundwater from the packed interval at a constant rate. Pumping rates were on the order of 2 to 20 gallons per minute (gpm).

Other methods used to define the water-yielding zones and vertical gradients were the salinity and temperature logs from the geophysical logging. This information was supplemented by a brine-tracing survey of the wells.

#### **2.5.4 Pumping Test**

A pumping test was performed in the GKM well to assess aquifer hydraulic characteristics and the suitability of the well for groundwater extraction. NPWA well S-9 was used to observe the drawdown. The time-drawdown data obtained during the test were analyzed using the Boulton delayed yield analytical method (Boulton, 1963).

The pumping test performed on the GKM well lasted 72 hours. The well was pumped from a depth of 145 feet. The pumping rate of the test varied from 16 to 13 gpm. The pump was a 5-horsepower Grundfos submersible pump with 1½-inch plastic discharge line. Flow was measured through a 1-inch totalizing flow meter and discharged to the sanitary sewer through two ¾-inch rubber hoses. Samples were collected from a brass valve at the surface, prior to the flow meter. The water level in the GKM well drew down approximately 82 feet during the pumping test. The water level in the observation well, S-9, drew down approximately 0.8 feet as a result of pumping at the GKM well.

### **2.6 Groundwater Sampling**

During the straddle packer tests in well S-9 and in the GKM well, groundwater samples were obtained from each packed interval and were submitted to the CLP or CRL for analysis for VOCs and TOC. These analyses provided information on the vertical distribution of contaminants.

Groundwater sampling was performed in NPWA wells S-8, S-9, and S-10; the GKM well; and private wells. Well sampling procedures varied depending on the type of well; these methods are discussed below. Groundwater samples were analyzed for

VOCs and TOC. Well sampling reports from the groundwater sampling are presented in Appendix D.

The wells already containing pumps were sampled by collecting the sample from a tap following at least 15 minutes of purging. Although pumping three well volumes was planned, homeowners requested that a shorter time be used to prevent wells from going dry. Field parameters were collected at each of the locations, except NPWA wells S-8 and S-10. NPWA wells S-8 and S-10 routinely pump continuously and are not specifically purged prior to sampling. NPWA well S-8 was sampled after wellhead treatment by chlorination because there was no way to collect a sample prior to treatment.

Wells in which pumps were installed for sampling (NPWA well S-9 and GKM well) were purged of at least three well volumes and until the field parameters stabilized. At the GKM well, the sample was collected after 24 hours of continuous pumping during the pumping test. The field parameters monitored were pH, conductivity, and temperature. These wells were purged with submersible pumps and sampled through the same pumps after the pumps were throttled back to less than 1 gpm. The pumps were decontaminated between uses. Purge water from well S-9 and the GKM well was discharged to the sanitary sewer. For all other wells, the purge water was discharged to the ground. Decontamination procedures were consistent with those outlined in the SAP.

All data on parameters measured during purging are provided in Appendix D.

The sampling round at the GKM well; NPWA wells S-8, S-9, and S-10; and private wells was performed after the initial sampling during the straddle packer tests. The residential wells selected for sampling were identified in the well inventory and are discussed in Section 3. All samples were analyzed for VOCs and TOC.

The Souderton Borough well located south of well S-9 was sampled during the GKM well aquifer test. This well was added to the sampling list late in the project and was sampled without purging well water. The sampling was accomplished by bailing a sample from about 30 feet below the water surface in the well.

Two dug wells in proximity of Gentle Cleaners were sampled. Several dug wells sampled during previous investigations or identified at the site during the well inventory were not selected for sampling in this investigation. These wells were not selected primarily because they were not near the PRPs. The water in all of these wells is from perching of water on top of the bedrock, not from the bedrock aquifer itself. The construction of these wells is very variable but generally consists of a stone-lined pit. The water in these wells is typically stagnant.

## **2.7 Close Support Laboratory**

### **2.7.1 Introduction**

A close support laboratory (CSL) was used during the investigation to rapidly and quantitatively screen soil and groundwater samples for various chlorinated organic compounds. The screening was accomplished by extracting the media using a heated headspace technique and analyzing the vapor samples with a field GC.

### **2.7.2 Objectives**

One objective of the CSL was to provide rapid field results for use in locating borings, selecting samples for laboratory analysis of soil samples, and monitoring of groundwater samples obtained during the aquifer pumping test. Another objective for the CSL was to determine if any of the compounds of interest were present in soil samples and to provide estimated concentrations of the compounds of interest. Estimated CSL data were used to guide field sampling by defining the extent and relative levels of contamination.

The methodology used to analyze samples in the CSL is discussed in detail in Appendix C.

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**Section 3**  
**Physical Characteristics of the Study Area**

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## Section 3 Physical Characteristics of the Study Area

### 3.1 Topography

The site is located in the Triassic Lowland Section of the Piedmont Physiographic Province. The topography of the area is gently rolling, with low-lying ridges and hills. The land and drainage in the vicinity of the site generally slopes to the southeast, toward the Delaware River. Most of the region around the site is drained by Skippack Creek and its tributaries; Skippack Creek then discharges into the Schuylkill River, which ultimately discharges into the Delaware River. Surface elevations vary from about 200 feet to about 600 feet above mean sea level (MSL). A topographic map of the site is provided in Figure 3-1.

The facilities investigated are located in urban areas and are generally flat. Fill materials were observed at each of the facilities.

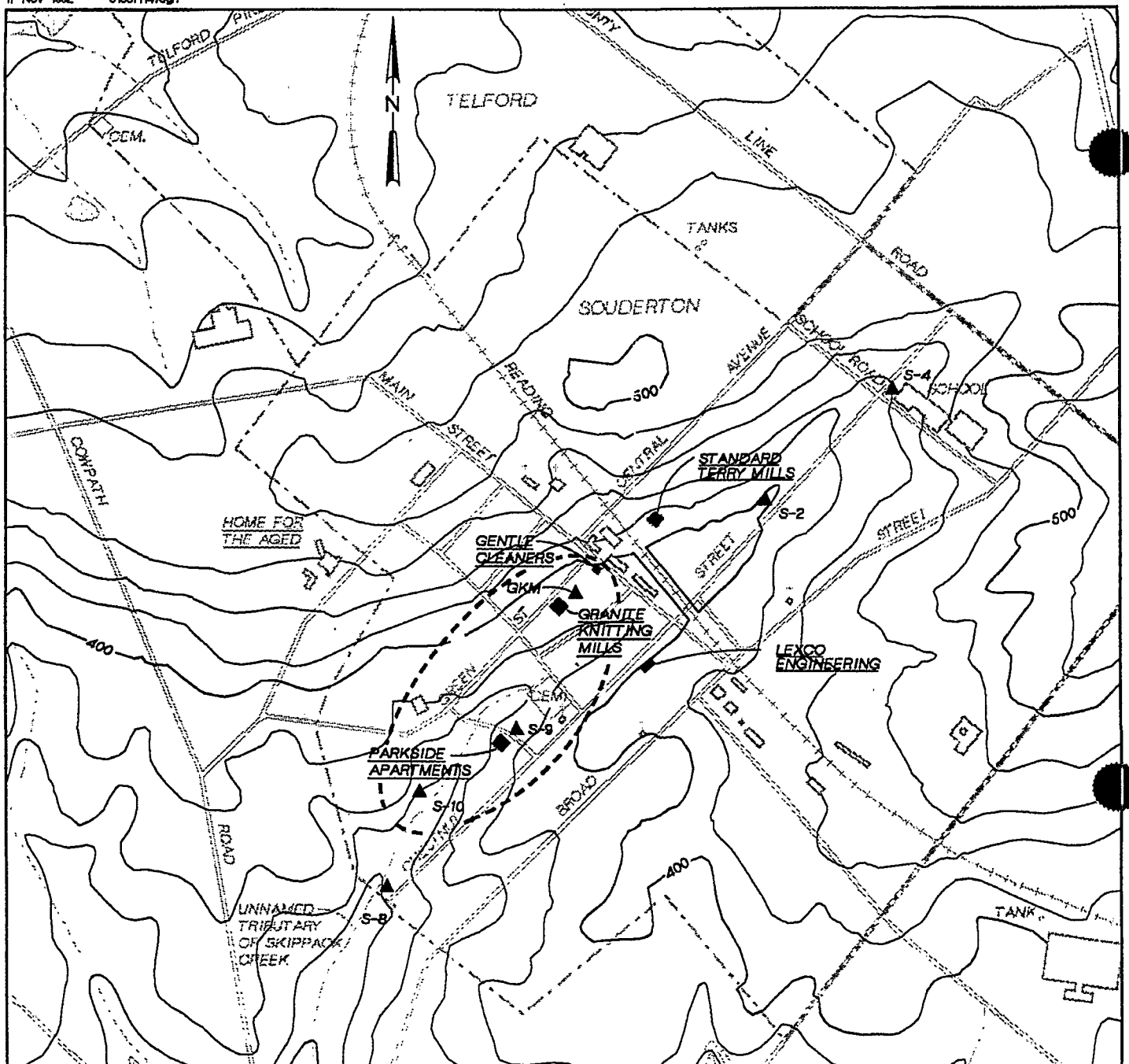
### 3.2 Climate

The climate of the area is moist, with moderate temperatures (Longwill and Wood, 1965; Newport, 1971). Most of the weather patterns are derived from the continental interior. Occasionally, coastal weather systems, which are characterized by severe storms with considerable rainfall and a potential for flooding, affect the area.

The average annual precipitation ranges from 43 inches in eastern Montgomery County to 47 inches in the northern part of the county. This precipitation is fairly well distributed throughout the year. The wettest month is usually August, while the driest month is typically October. Most of the annual precipitation falls during spring and summer. The total annual precipitation includes 20 to 30 inches of snowfall, and snow covers the ground about one-third of the time during winter.

Temperatures range from zero to 100°F. The long summers are characterized by daily high temperatures in the 90s. The winters are mild, and the minimum temperatures fall below freezing an average of fewer than 100 days each year. The average temperature is about 54°F, with monthly averages ranging from 33°F (the average for February) to 77°F (the average for July). The freeze-free season typically ranges from 170 to 200 days.

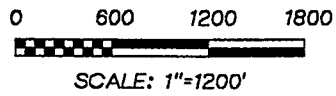
The mean annual rate of evaporation from surface-water bodies in the county is 33 inches. However, because the free-water surface area is so small, transpiration by vegetation constitutes the primary means of returning water to the atmosphere. About 26 inches of precipitation are returned to the atmosphere by the processes of evaporation and evapotranspiration (Longwill and Wood, 1965; Newport, 1971).



SOURCE: U. S. GEOLOGICAL SURVEY 7-1/2 MINUTE QUADRANGLE MAP FOR TELFORD, PA.

**LEGEND**

- ▲ WELL LOCATION
- BOROUGH AND TOWNSHIP BOUNDARIES
- 400— TOPOGRAPHIC CONTOUR (ELEVATION IN FEET ABOVE MSL)
- POTENTIALLY RESPONSIBLE PARTY (PRP)



**FIGURE 3-1**  
**SITE TOPOGRAPHY**  
 North Penn Area 1  
 Phase II RI/FS



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### 3.3 Surface Water Hydrology

Apart from the United States Geological Survey (USGS) topographic map (Telford, Pennsylvania, 7-1/2 minute quadrangle map), a detailed description of the surface-water hydrology is not available. According to Newport (1971), an estimated 15 to 21 inches of precipitation enters the surface-water drainage system as surface runoff. In the vicinity of the site, it appears that surface runoff moves toward the unnamed, intermittent tributary of Skippack Creek (Figure 3-1), which flows through the site; some runoff may be directed elsewhere by stormwater collection systems. When the water table is high, water entering this stream may flow southwestward and southward into Skippack Creek and then to the Schuylkill and the Delaware rivers. When the water table is low, the surface runoff may seep into the dry stream bed and emerge at the surface farther downstream. The intermittent tributary to Skippack Creek was observed to contain water for the duration of site activities, which occurred in the late spring and early summer.

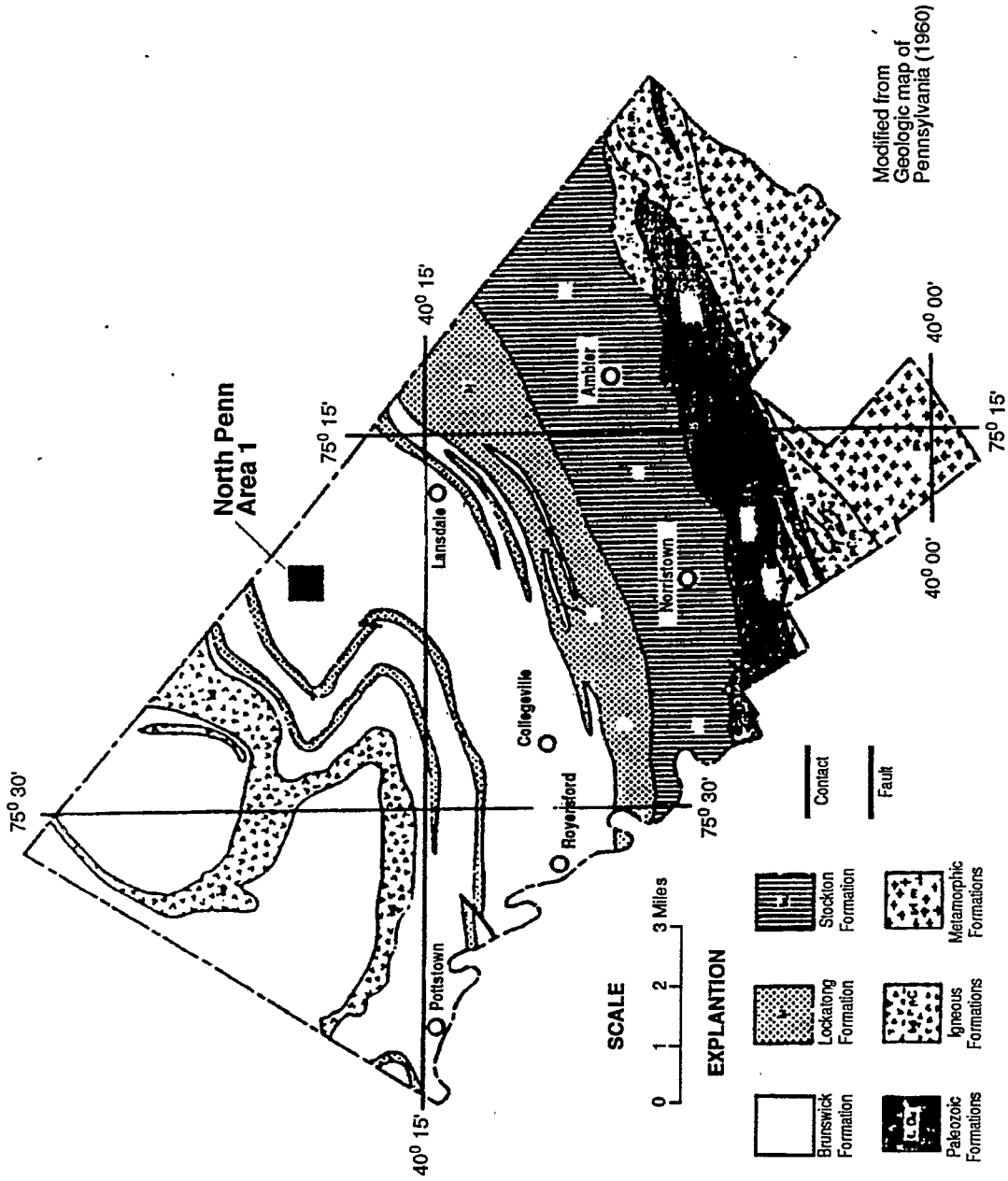
### 3.4 Geology

The rocks underlying the area around the site are typically composed of the Triassic deposits of the Newark Basin (Figure 3-2) (Longwill and Wood, 1965; Newport, 1971). A generalized geologic section for Montgomery County is presented in Table 3-1 (Newport, 1971).

The principal bedrock unit is the Brunswick Formation. This formation consists of thin, discontinuous beds of reddish-brown shale interbedded with mudstone and siltstone. The rocks are composed chiefly of feldspar, illite, chlorite, quartz, and calcite. The total thickness of the Brunswick near the site is on the order of 9,000 feet (Newport, 1971).

The Brunswick is underlain by the Lockatong Formation, and in some areas the two formations interfinger. The Lockatong consists of massive beds of medium and dark gray argillite interbedded with thin beds of gray-to-black shale and siltstone. Some dolomite, feldspar, clay, and quartz are present. The Lockatong is more resistant to erosion and forms a low ridge when outcropping at the surface. The maximum thickness of the Lockatong in the vicinity of the site is about 4,000 feet (Newport, 1971).

The Stockton Formation underlies the Lockatong and consists of interbedded layers of sandstone and shale. The formation is typically divided into three members. The lower member is characterized by red-to-gray medium- to coarse-grained arkosic sandstone and conglomerate. Numerous lenses of silty and sandy red shale are interbedded with the sandstone. The middle member consists of brown, red, and gray fine- to medium-



Source: Newport, 1971

**Figure 3-2**  
**GENERALIZED GEOLOGIC MAP**  
 North Penn Area 1  
 Phase II RI/FS

**Table 3-1**  
**GEOLOGIC SECTION FOR MONTGOMERY COUNTY, PENNSYLVANIA**  
 (from Newport, 1971)

Era, System, and Epoch	Formation	Thickness (feet)	Character
Cenozoic			
Quaternary			
Holocene	Alluvium	0-10	Soil, sand, gravel, and clay; deposits in stream valleys.
Pleistocene	Pensauken Formation	0-10	Sand, gravel, clay, yellowish-brown; small areal extent.
Tertiary			
Pliocene (?)	Bryn Mawr Gravel	0-10	Sand and gravel; small areal extent.
Mesozoic			
Cretaceous	Patapsco Formation	0-10	Clay and sand, highly colored; small areal extent.
Triassic	Diabase	5-1,800	Medium- to coarse-grained igneous rock, dark gray, occurs as dikes and sills.
	Brunswick Formation	9,000-16,000	Shale, mudstone, sandstone, and conglomerate beds; reddish-brown.
	Lockatong Formation	0-2,000	Argillite, mudstone, and shale; dark gray to black, thick bedded.
	Stockton Formation	1,000-6,000	Shale and siltstone in upper member; sandstone, fine- to coarse-grained, arkosic, middle member; conglomerate lower member.
Paleozoic			
Ordovician	Conestoga Limestone	500-800	Limestone, impure, thin-bedded upper part; middle dark graphitic phyllite, lower limestone, granular thick-bedded, dark gray.
Cambrian	Elbrook Formation	800	Limestone, fine-grained, light gray to cream-colored, thin-bedded.
	Ledger Dolomite	1,000	Dolomite, granular, gray to bluish gray.
	Harpers Formation	500-800 (?)	Phyllite, fine-grained, greenish-gray, some beds of quartzite and schist.
	Chickies Quartzite	500-1,000	Quartzite, vitreous, light-colored thick-bedded, conglomerate at base.
Precambrian (?)	Wissahickon Formation	---	Schist (albite-chlorite and oligoclase-mica), includes hornblende gneiss and phyllite.
Precambrian	Granite gneiss	---	Composed chiefly of quartz, feldspar, biotite, and hornblende.
	Hornblende gneiss	---	Composed of quartz, feldspar, and hornblende.
	Serpentine	---	Soft, fine-grained, green.

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grained arkosic sandstone with thick beds of red shale and siltstone. The sandstones of this member are more well-sorted than the sandstones of the lower member. The upper member comprises very fine-grained arkose and siltstone with an extremely hard and resistant layer of red and gray shale at the top. The total thickness of the Stockton in the vicinity of the site is about 6,000 feet (Newport, 1971).

Diabase dikes and sills occur in the subsurface and are exposed at the surface in some parts of Montgomery County. These features are composed of very dense fine-grained black diabase containing primarily augite and labradorite. The dikes vary from 5 feet to 100 feet in thickness, and the sills may exceed 1,000 feet in thickness at some locations (Newport, 1971).

The sedimentary formations typically dip to the northwest and the north at an average angle of about 20 degrees and strike approximately northeast-southwest. Several broad anticlines and synclines have been identified.

Most of the rocks in the vicinity of the site are cut by a well-developed system of nearly vertical joints. Three distinct joint sets have been identified in the Brunswick (JACA, 1987). One set strikes north-northeast while the other sets are reportedly less well-developed and strike northwest and east-northeast. All three joint sets are nearly vertical in dip and are separated by an average distance of about 6 inches. These joints are common in the Brunswick but are narrower and more widely spaced in the Lockatong. Where the Brunswick and Lockatong are interfingered, the rocks are characterized by a greater number of fractures. Joints in all formations are generally partly filled with either quartz or calcite cement (horizontal bedding plane jointing).

As part of this RI, in order to evaluate the distribution of fractures at and in the vicinity of the site, a field investigation and a review of aerial photographs were performed. The field investigation involved locating rock outcrops and measuring the strike, dip, and frequency of fractures. The investigation also included field verification of fracture trends identified from aerial photographs. The review of aerial photographs covered an area out to approximately 3 miles around the site.

During the RI, fractures were identified at the site in the stream bed of the unnamed tributary to Skippack Creek; this was the only rock outcrop that could be identified within 1 mile of the site. Three sets of fractures were identified in the stream bed. Two of these sets of fractures were observed to be almost vertical and were oriented east-northeast and north-northwest. An additional set of fractures was observed oriented north-northeast and dipping approximately 30 degrees to the west; this set may represent the bedding planes of the bedrock units. The third set of fractures was less well developed than the other two. The fracture spacing was observed to range from 2 inches to 3 feet at this location.

Some faulting reportedly has occurred in the vicinity of the site. The faulting is typically normal in nature with the upthrown block on the north or west side. The

Chalfont Fault trends in an east-west direction about 1 mile south of the site. Vertical offset along this fault is about 3,000 feet (Willard, et al., 1959). Two drag faults that trend northeast to southwest occur just north of this fault; a portion of the Lockatong is offset by these smaller faults.

As part of this RI, a fracture-trace analysis was performed with aerial photographs of the site. Stereographic pairs were used to identify potential fractures. The potential fractures were then checked in the field. Figure 3-3 shows fracture traces identified within 1 mile of the site. This fracture-trace set was oriented northwest, and during field checks was characterized as stream valleys.

Other fracture traces identified in the vicinity of the site, but more than one mile from the site, are discussed below:

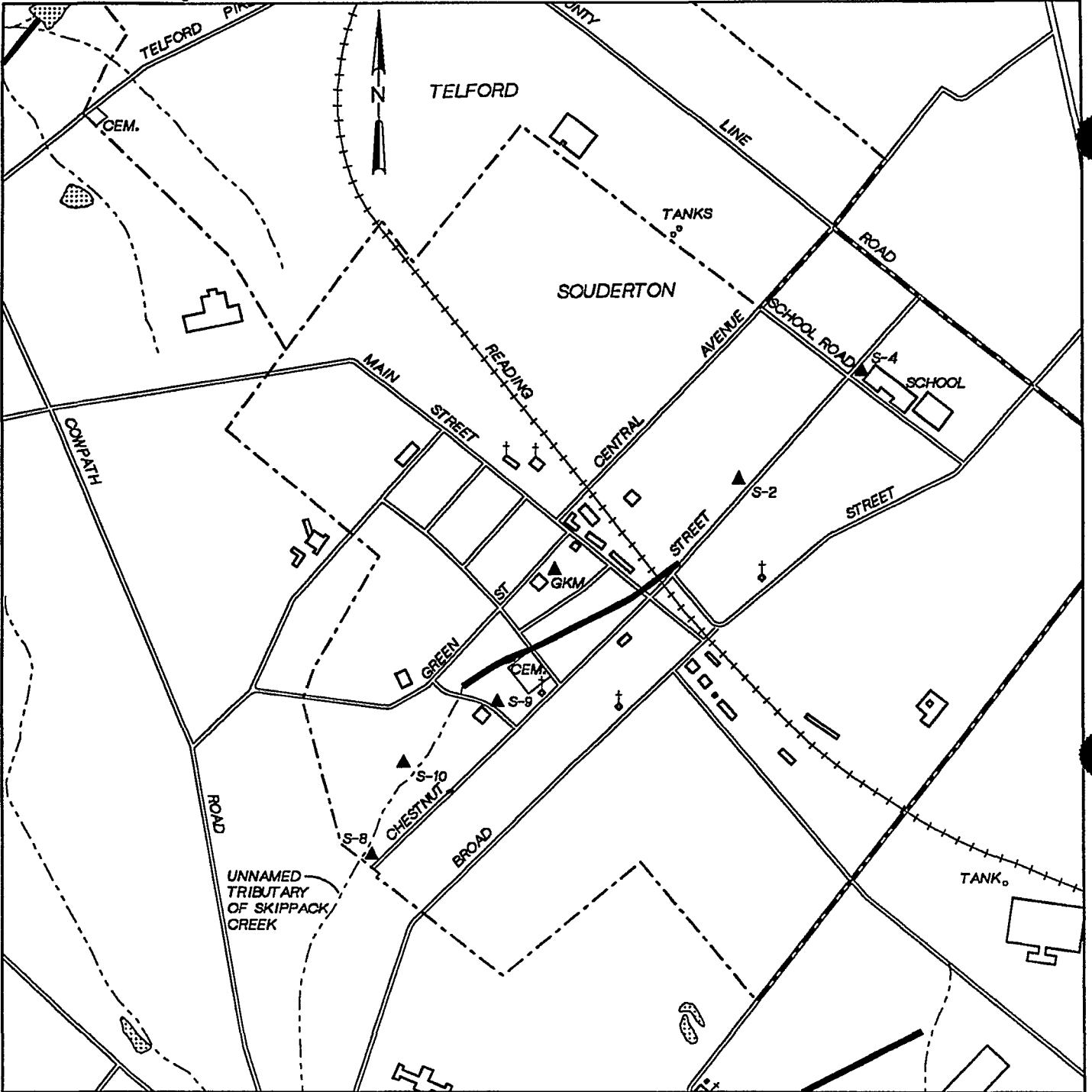
- A fracture-trace set was observed oriented north-northwest. The fractures are characterized as stream valleys. The fractures are generally 600 to 700 feet long and were observed in the field to be present, but not strong.
- A fracture-trace set was noted oriented east-northeast. These fractures were long, in excess of a mile in some cases. The surface expression of these fractures is as slopes changing the elevation of the fields. This fracture set includes the Chalfont Fault.

Figure 3-4 is a rose diagram of all fracture orientations determined from aerial photographs and field observations. The north-northwest, north-northeast, and east-northeast trends are apparent.

### 3.5 Soil

The soils at the site belong to several soil groups (SCS, 1986). In the immediate vicinity of Gentle Cleaners and Granite Knitting Mills is "made" land, which is land resulting from the altering and mixing, by construction activity, of soils formed in material weathered from shale and sandstone. The soils are typically shaley silt loam to sandy loam, although in some areas the soils consist entirely of pieces of shale. Depth to bedrock may be as much as 6 feet. Estimated permeability ranges from moderate to very slow.

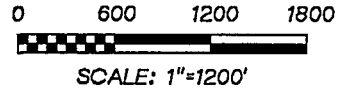
Soil samples were collected at the Gentle Cleaners, Granite Knitting Mills, and Parkside Apartments for geotechnical analysis. The geotechnical analyses results are presented in Appendix E. The results of the geotechnical analyses indicate that the



SOURCE: U. S. GEOLOGICAL SURVEY 7-1/2 MINUTE QUADRANGLE MAP FOR TELFORD, PA.

**LEGEND**

- ▲ WELL LOCATIONS  
( S-X = NPWA WELL  
GKM = GRANITE KNITTING MILLS WELL)
- POTENTIAL FRACTURE TRACE



**FIGURE 3-3**  
**POTENTIAL FRACTURE TRACES**  
 WITHIN 1 MILE OF THE SITE  
 North Penn Area 1  
 Phase II RI/FS

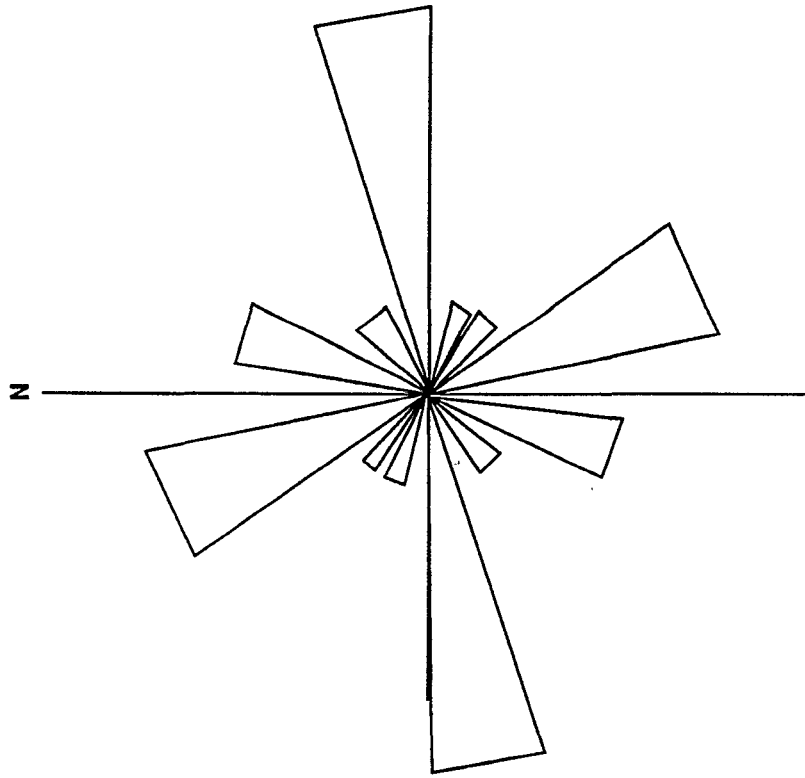


AR300711





**Figure 3-4**  
**ROSE DIAGRAM OF FRACTURE ORIENTATIONS**  
North Penn Area 1  
Phase II RI/FS



soils are primarily silts with traces of coarser and finer materials. The soils exhibited low plasticity and very low permeability (ranging from 0.012 to 0.020 feet per day). Total porosity ranged from 0.36 to 0.41. The soils were found to be almost saturated.

Toward the southwestern portion of the site, undisturbed soils occur. These include Abbottstown silt loam and Rowland silt loam. The Abbottstown silt loam occurs on broad undulating and smooth uplands, and the depth to bedrock is typically within 2 to 3 feet of the surface. The Rowland silt loam occurs in narrow bands along most streams. Other undisturbed soils in the site vicinity include Readington silt loam, Reaville shaley silt loam, Klinesville shaley silt loam, and Klinesville very shaley silt loam. These soils are characterized by depths to bedrock of less than 5 feet.

The soil samples collected at each facility were logged using the Unified Soil Classification System. The soils at each facility are discussed below.

- At Gentle Cleaners, the soils were moderate yellowish brown silts with little clay and a trace gravel. The soils were moist at the surface, and deeper soils were dryer. Organic materials were noted in the surface soils. The soils were approximately 10 feet thick in the cleaners' backyard, with some fill noted. The soils in the adjacent backyard were approximately 5 feet thick due to their lower elevation. Degraded fractured bedrock was located below the soils.
- At Granite Knitting Mills, the soils were observed to be moderate brown fine sandy silts, with variable sand, silt, clay, and shale fragments. The soil moisture was variable. The soils were covered with asphalt. Thickness of soils ranged from 6 to 13 feet but were generally thinner in the alley south of the facility. Fill materials were noted sporadically.
- At Parkside Apartments, the soils were observed to be moderate yellowish-brown, gravelly, clayey silt. Shale fragments were common. These soils were wet, and grass covered the facility. Soil thickness ranged from 5 to 12 feet. The soils appeared to be fill near the unnamed tributary to Skippack Creek.
- At Lexco Engineering, the soils were red-brown shale and clay mixtures. The soils were moist and covered by gravel fill. Soil thickness ranged from 4 to 7 feet. An ash layer in some of the borings indicates that the area has been filled.
- At the former Standard Terry Mills, the soils were red-brown clayey silt, with shale fragments. The soils were covered with extensive fill, generally characterized as construction debris. The facility was intermittently covered by asphalt.

At each of the facilities, soils were noted as moist in some of the borings. Because the geotechnical analyses indicate that the soils contain a significant percentage of silt and clay particles, the possibility exists that the soils may become saturated and that perched water conditions may exist, probably varying seasonally. Water was not observed in any of the boreholes, but water movement may be too slow to have filled the borings during drilling. The presence of dug wells in the area also suggests that perched water conditions may exist in the soils above bedrock.

## **3.6 Groundwater**

### **3.6.1 Regional Hydrogeology**

According to Longwill and Wood (1965), the Brunswick Formation is considered to be a reliable source of small to moderate supplies of groundwater. Their analysis of almost 200 wells in Montgomery and Berks Counties indicated that wells should be installed to a depth of at least 200 feet if yields in excess of 100 gpm are desired. Typically, wells drilled to between 200 feet and 550 feet provided maximum yields. The more site-specific information provided by Newport (1971) only weakly demonstrates this relationship. Table 3-2 provides well construction and yield data on several wells in the vicinity of the site; the locations of all of these wells are shown in Figure 3-5. In general, it appears that wells shallower than 200 feet or deeper than 550 feet have lower yields than those in between, but wells drilled to depths between 200 feet and 550 feet have greatly varying yields, with some well yields below 50 gpm.

The greater yields observed in the wells drilled to depths between 200 feet and 550 feet may be misleading. For example, well NP-69 was drilled to a depth of 500 feet and has a maximum short-term yield during summer conditions of about 200 gpm. Four distinct water-bearing zones were encountered during the drilling of the well, at depths of 134, 337, 355, and 405 feet. These zones contributed 171, 31, 5, and 16 gpm, respectively, to the yield of the well (JACA, 1987). In other words, almost 80 percent of the total yield was contributed by a water-bearing zone within 200 feet of the surface. Similar conditions were encountered in the NPWA's wells L-8 and L-9 in Lansdale.

### **3.6.2 Site Hydrogeology**

The results of the video and geophysical logging performed as part of this RI are summarized in Figures 3-6 and 3-7. Fracture zones identified in the video logs are shown in the next-to-last column of each figure. Intervals shaded in gray have large numbers of fractures within the shaded interval. Fractures were widely scattered throughout the depth of the GKM well but were concentrated in the upper portion of well S-9. Horizontal fractures identified in video logging may be bedding plane fractures.

Table 3-2  
WELL INVENTORY

Well ID	Well Name/ Location	Onsite/ Offsite	Use	Depth (feet)	Construction	Depth to bottom of Casing (feet)	Diameter (Inches)	Yield (gpm)	Age
R-1	250 Green St.	off	potable	--	--				
R-2	311 Cowpath Rd.	off	potable	250	drilled				30
R-3	320 Cowpath Rd.	off	potable	90	drilled				40
R-4	240 Green St.	off	potable	200	drilled				30
R-5	113 Penn Ave.	off	outside	--	dug				
R-6	66 Walnut St.*	off	outside	--	dug				
R-7	332 W. Broad St.	off	potable	120	drilled				50
R-8	335 W. Broad St.	off	potable	100	drilled				50
R-9	127 W. Chestnut St.	on	outside	--	--				70
R-10	Mennonite Homes	off	outside	192	drilled				
R-11	Walnut & Wile	off	--	--	dug				
R-12	302 Main St.	off	--	--	dug				
R-13	23 Green St.	on	not in use	--	dug				
R-14	27 Green St.	on	not in use	--	dug				
R-15	143 Green St.	on	abandoned	--	dug				
R-16	161 Main St.	on	not in use	--	dug				
R-17	160 Main St.	on	not in use	--	dug				
R-18	112 W. Chestnut	on	--	--	dug				
R-19	14 Green St.	on	outside	--	dug				
R-20	18 Green St.	on	not in use	--	dug				
GKM	Granite Knitting Mills	on	not in use	208	drilled	10	8-6	11	1957
679	Souderton Borough Well	on	not in use	308	drilled	--	12-8	250	1961

Table 3-2  
WELL INVENTORY

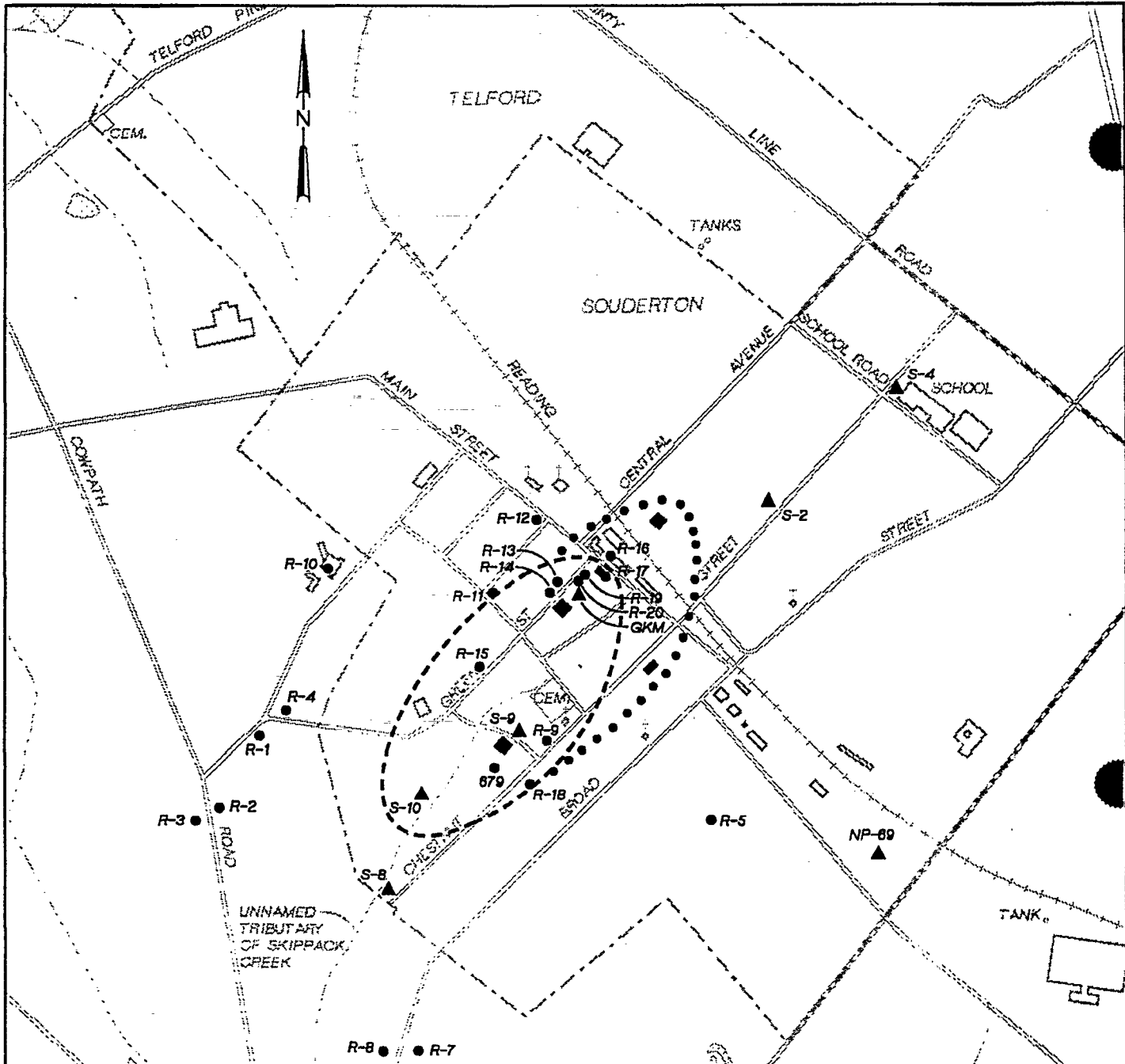
Well ID	Well Name/ Location	Onsite/ Offsite	Use	Depth (feet)	Construction	Depth to bottom of Casing (feet)	Diameter (inches)	Yield (gpm)	Age
S-10	NPWA S-10	on	public water supply	400	drilled	30	8	45	
S-2	NPWA S-2	off	public water supply	205	drilled	25	12	80	1960
S-4	NPWA S-4	off	public water supply	300	drilled	13	10	45	1953
NP-69	NP-69	off	public water supply	500	drilled	20	8	200	

-- unknown

\*Location corresponds with R-11 at the corner of Walnut and Will Streets—may be same well.

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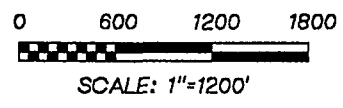
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SOURCE: U. S. GEOLOGICAL SURVEY 7-1/2 MINUTE QUADRANGLE MAP FOR TELFORD, PA.

**LEGEND**

- ▲ S-2 NPWA WELL
- R-X RESIDENTIAL WELL
- ▲ GKM GRANITE KNITTING MILLS WELL
- BOROUGH AND TOWNSHIP BOUNDARIES
- - - PRELIMINARY BOUNDARY OF AREA 1 (NUS, 1986a)
- ..... REVISED BOUNDARY OF AREA 1 (1991)
- POTENTIALLY RESPONSIBLE PARTY (PRP)



**FIGURE 3-5**  
**WELL LOCATION MAP**  
North Penn Area 1  
Phase II RI/FS



AR300717

LOG TYPE							
DEPTH BELOW GROUND SURFACE AND ELEV. MSL surface 385.9	PEAK VALUES			FLUID RESISTIVITY (ohm * m)	FLUID TEMPERATURE (°C)	FRACTURE ZONES (FROM VIDEO LOG)	PACKER INTERVALS
	SINGLE POINT ELECTRIC <sup>a</sup> (ohms)	NATURAL $\gamma^b$ (counts/sec)	BOREHOLE DIAMETER <sup>c</sup> (in)				
350		470	10				
			18				
		350	12				
1150			9-10				
			9	6264	14.07		
1080		350			14.07		
1020			10	6156	14.08		
1020		730					
		400	9-11	5973	14.06		
300		1020	9				
				5948	14.06		
		1020					
		1080	9	6010	14.04		
		350	9	5805	14.00		
1080		420	14	5634	13.97		
		1150					
		420		5633	13.91		
250		1350	7-8				
		430			13.90		
		1200					
		430		5645	13.82		
		430	7				
		420	7	5632	13.77		
150		1000					
		420		5606	13.73		
		1000					
		510		5534	13.69		
200		1040					
		420					
		1240					
		390		5568	13.64		
		430					
		510		5565	13.59		
		1440					
		1260	7				
				5560	13.57		
200		430			13.55		

**Note:**

Shaded areas on single point electric and natural gamma logs represent transition zones to the indicated peak value

<sup>a</sup>Background is 800 ohm

<sup>b</sup>Background is 300

<sup>c</sup>Drilled hole is 8" to 100 ft., then 6"

**Figure 3-6**

GKM WELL - GEOPHYSICAL LOGS

North Penn Area 1

Phase II RI/FS



AR300718

DEPTH BELOW GROUND SURFACE AND ELEV. MSL surface 355.94	LOG TYPE						
	SINGLE POINT ELECTRIC <sup>a</sup> (ohms)	NATURAL $\gamma^b$ (counts/sec)	BOREHOLE DIAMETER <sup>c</sup> (in)	FLUID RESISTIVITY (ohm * m)	FLUID TEMPERATURE (°C)	FRACTURE ZONES (FROM VIDEO LOG)	PACKER INTERVALS
350		>1,000					
50	850		11	5253	12.51		
300		440	11	5180	12.66		
	900	>1,000		5202	12.69		
				5285	12.70		
		400	9	5375	12.68		
100	850			5441	12.70		
250		460	9-13	5488	12.70		
	950		11-13		12.70		
			9-11	5553	12.71		
				5577	12.71		
150	900		9-10	5596	12.69		
200	880			5612	12.72		
		450	9	5625	12.70		
	1000			5638	12.71		
				5649	12.71		
200		>500		5659	12.73		
150	940	>>500	12	5691	12.73		
		470		5706	12.74		
			9	5712	12.82		
	960			5717	12.87		
			16	5694	12.89		
250	850	450		5664	12.91		
100	900		9	5640	12.91		
			9	5640	12.91		
	920		9	5595	12.91		
		>1,000		6288	12.89		
					12.88		
300	>1,000			6092			
					12.85		

**Note:**

Shaded areas on single point electric and natural gamma logs represent transition zones to the indicated peak value

- <sup>a</sup>Background is 900 ohm
- <sup>b</sup>Background is 300
- <sup>c</sup>Drilled hole is 8"

**Figure 3-7**  
**WELL S-9 - GEOPHYSICAL LOGS**  
 North Penn Area 1  
 Phase II RI/FS





The single point electric and fluid resistivity logs were evaluated to identify potential changes in strata. In the GKM well, the peaks on the single point electric log and deflections on the fluid resistivity logs do not suggest strata changes. Natural gamma logs were used to suggest clay content. Because the lithology of the Brunswick and Lockatong formations is very similar and natural gamma logs are sensitive to borehole diameter, the natural gamma logs were difficult to interpret. In comparing natural gamma radiation at the GKM well to the single point electric and fluid resistivity logs, the responses could not be used to interpret the geology.

In the S-9 well, the fluid resistivity log was very smooth indicating fairly homogeneous geology. The natural gamma logs in the S-9 well correlated with the fracture zones identified during video logging.

The single point electric, fluid resistivity, natural gamma, and temperature logs provided little or no usable information. The borehole diameter as defined by the caliper log correlated well with the intervals containing fractures indicated by the video log. In the GKM well, fluid resistivity dropped abruptly at a depth of 60 feet, indicating that fresh, low-conductivity water was entering the well at this depth from the fracture noted in the video log. The resistivity in this well generally declined with depth as more fresh water entered the borehole. In general, the fluid resistivity in well S-9 increased with depth, consistent with the concentration of fractures in the upper part of the borehole.

The temperature logs in the two wells are consistent with observed fracture distributions. The decrease in fluid resistivity in the GKM well correlates with a decrease in fluid temperature. The higher concentration of fractures in the S-9 well in the upper portions correlates with an increase in fluid temperatures with depth. No zones of anomalous temperatures were noted on the logs.

Brine tracing was performed in selected intervals of each well. The intervals were selected based on fractures identified from the caliper logs and variations in the fluid resistivity. In all cases, the tracer testing suggested that there was little or no vertical flow in either of the wells.

Copies of the geophysical logs provided by the USGS are included in Appendix B. A summary of the downhole television logging performed at the site also is included in Appendix B.

The packer testing was performed in each well in the zones identified in Figures 3-6 and 3-7. The packer testing indicated that the yield in each well was from multiple fracture zones. The GKM well had a combined yield of approximately 20 gpm. The individual zones were found to yield between less than 2 gpm and about 10 gpm (Table 3-3). At well S-9, the individual zones were found to yield from 8 to more than 16 gpm.

**Table 3-3  
PACKER SAMPLING ZONE YIELDS**

<b>Zone</b>	<b>Depth (ft)</b>	<b>Yield (gpm)</b>
GKM S-3	0 - 28	N/A
GKM S-2	60 - 80	6
GKM S-4	90 - 110	10
GKM S-5	110 - 130	4
GKM S-1	165 - 189	<2
S-9 S-1	77 - 97	10
S-9 S-2	130 - 150	>20
S-9 S-3	158 - 178	8
S-9 S-4	195 - 215	>15
S-9 S-5	245 - 265	16

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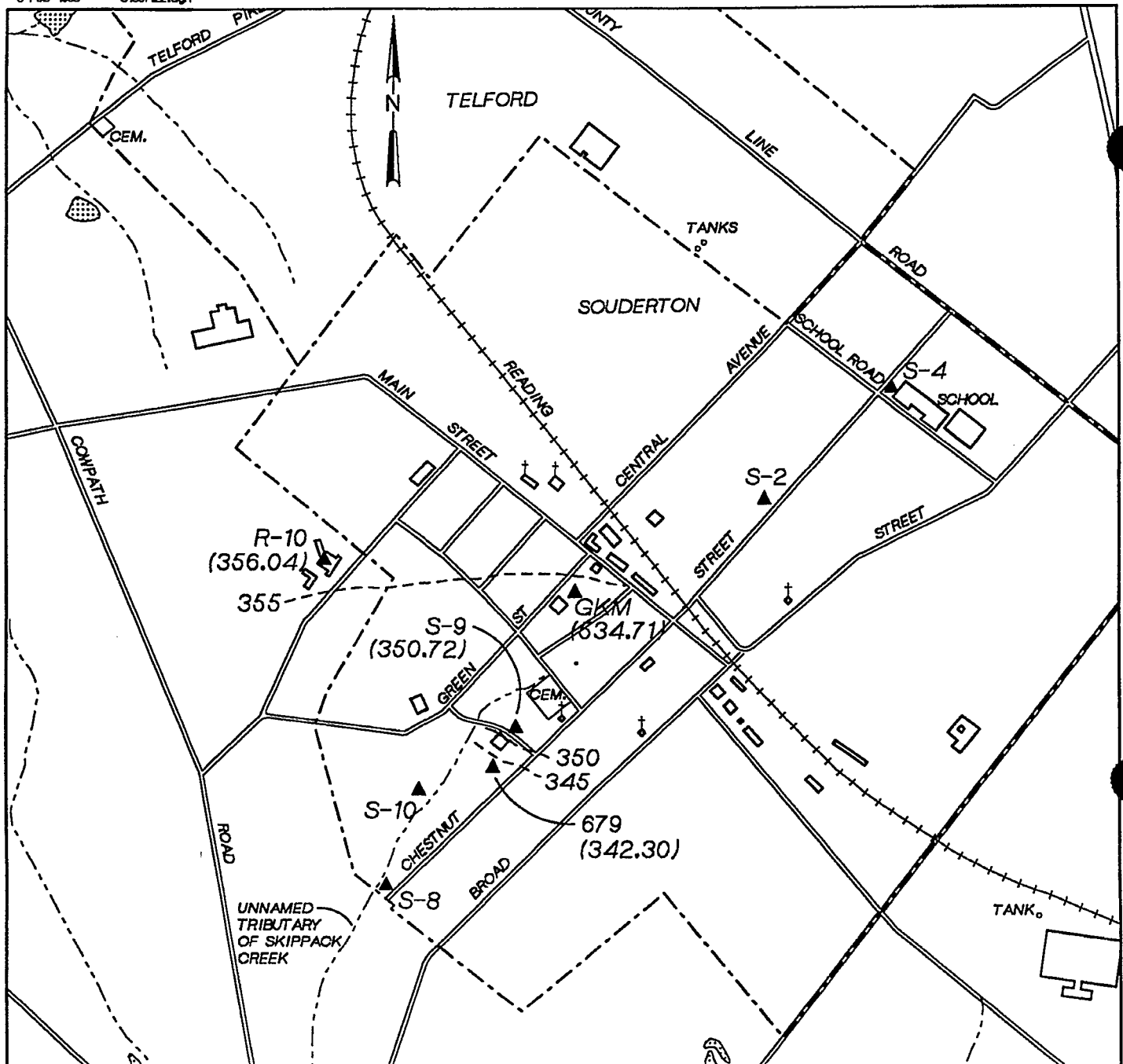
Only four bedrock aquifer wells were identified in site activities. A potentiometric surface map was generated from the water levels collected from each of these wells. The potentiometric surface map is Figure 3-8. Although the potentiometric surface is not well defined, it is consistent with site topography and indicates that groundwater is moving to the south, generally toward the pumping centers at NPWA wells S-10 and S-8. Reports from NPWA personnel indicate that the unnamed tributary to Skippack Creek is a losing stream, probably because of the pumping at the NPWA production wells.

During earlier investigations, aquifer tests were conducted in the vicinity of the site to provide estimates of the hydraulic characteristics of the water-bearing units. Longwill and Wood (1965) report a test in which well 665 (NPWA well S-9) was pumped for 69 hours at a rate of 150 gpm in March 1961. The transmissivity was estimated at 1,200 ft<sup>2</sup>/day using data from well 679, and at 470 ft<sup>2</sup>/day using data from the pumping well. Well 679, the Souderton Borough Well, is located approximately along the strike of the bedrock from NPWA well S-9. A pumping test also was performed in well NP-69 for 48 hours at a rate of 174 gpm in July 1986. This test provided a transmissivity estimate of 60 ft<sup>2</sup>/day.

Each of these tests was analyzed using methods designed for isotropic, homogeneous, and porous media. Because of the fractured nature of the bedrock in the vicinity of the site, this assumption may be only partially met. Longwill and Wood (1965) discuss other tests in the region where observation wells were available both along and across the strike of the rocks. Typically, the drawdown was less in the observation well across the strike than in the observation well along the strike, indicating that the latter is a preferred trend of fracture development.

A 72-hour pumping test was performed, as part of this RI, on the GKM well. The pumping test was conducted to determine if pumping at the GKM well would affect the NPWA well S-9. The pumping test was conducted at a pumping rate of 16 gpm initially; the rate dropped gradually throughout the test to 13.5 gpm. The pumping was performed with a submersible pump installed at a depth of 145 feet in the well; the total depth of the well based on the video logging was 189 feet. The water level in the pumping well was drawn down from a depth of 35 feet to a final depth of 120 feet. The pumping test was monitored in the pumping well and at the NPWA well S-9. A data logger and pressure transducer were used in the pumping well. A data logger and a pressure transducer also were set up in well S-9 four days prior to beginning the pumping test and remained in the well for 2 days following the pumping test.

The data collected at the NPWA well S-9 are plotted in Figure 3-9. The beginning and ending of the test are indicated on the figure. The data indicate that the pumping at the GKM well had a definite influence on the water level in well S-9. There was a downward trend, probably a seasonal decline, in the water level in well S-9 before initiation of pumping at the GKM well; the linear trend was removed for plotting, Figure 3-9.



SOURCE: U. S. GEOLOGICAL SURVEY 7-1/2 MINUTE QUADRANGLE MAP FOR TELFORD, PA.

**LEGEND**

- ▲ WELL LOCATION SYMBOL:  
R-X = RESIDENTIAL WELLS  
GKM = GRANITE KNITTING MILLS WELL  
S-X = NPWA WELLS

----- POTENTIOMETRIC SURFACE CONTOUR  
(FEET ABOVE MSL)

(342.30) WATER LEVEL ELEVATION  
(FEET ABOVE MSL)

0 600 1200 1800

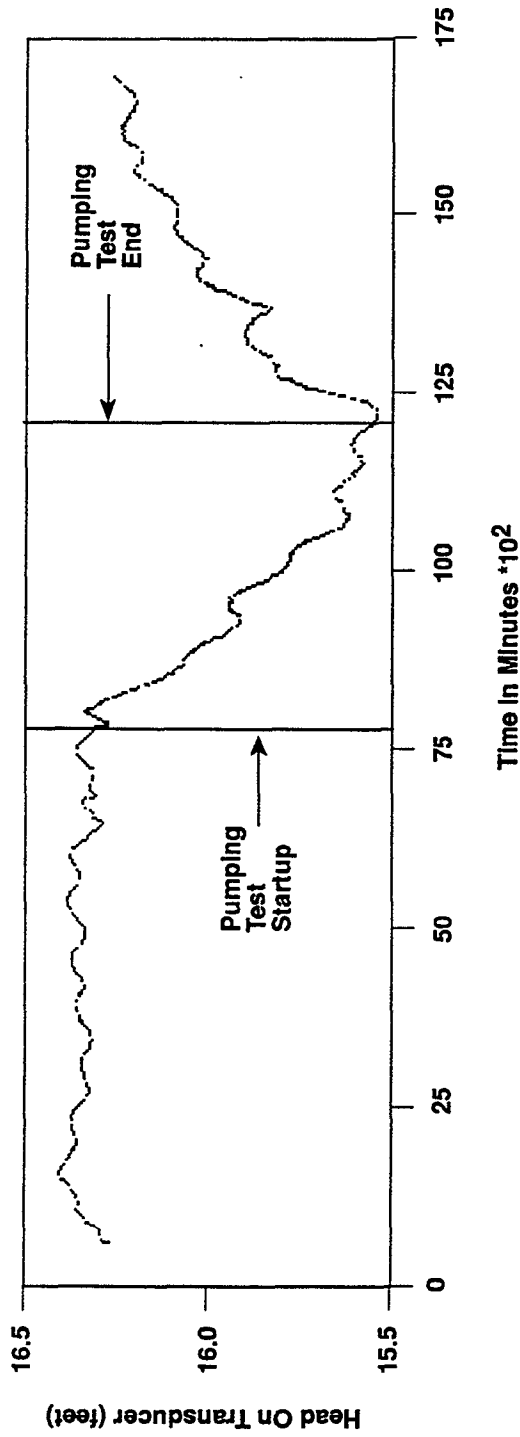


SCALE: 1"=1200'

**FIGURE 3-8**  
POTENTIOMETRIC SURFACE MAP  
North Penn Area 1  
Phase II RI/FS



AR300723



**Figure 3-9**  
WATER LEVELS IN WELL S-9 CORRECTED  
FOR BACKGROUND WATER LEVEL DECLINE  
North Penn Area 1  
Phase II RI/FS



The drawdown in well S-9 was separated from the general downward trend in water level and is plotted on log-log scales in Figure 3-10. This plot shows that the drawdown in the monitoring well for the pumping test had not stabilized before the pumping test was ended.

Due to the variability in the water level in the well and the fact that drawdown was still occurring when the test was terminated, it was difficult to fit the data to a type curve using the Boulton (1963) method. Although a range of match points was possible, the match point shown on the figure is approximately representative. The match points were substituted into the equations:

$$T = \frac{Q}{4\pi S} W(u_{AY}, r/B)$$

and

$$S = \frac{4Tt u_A}{r^2}$$

where

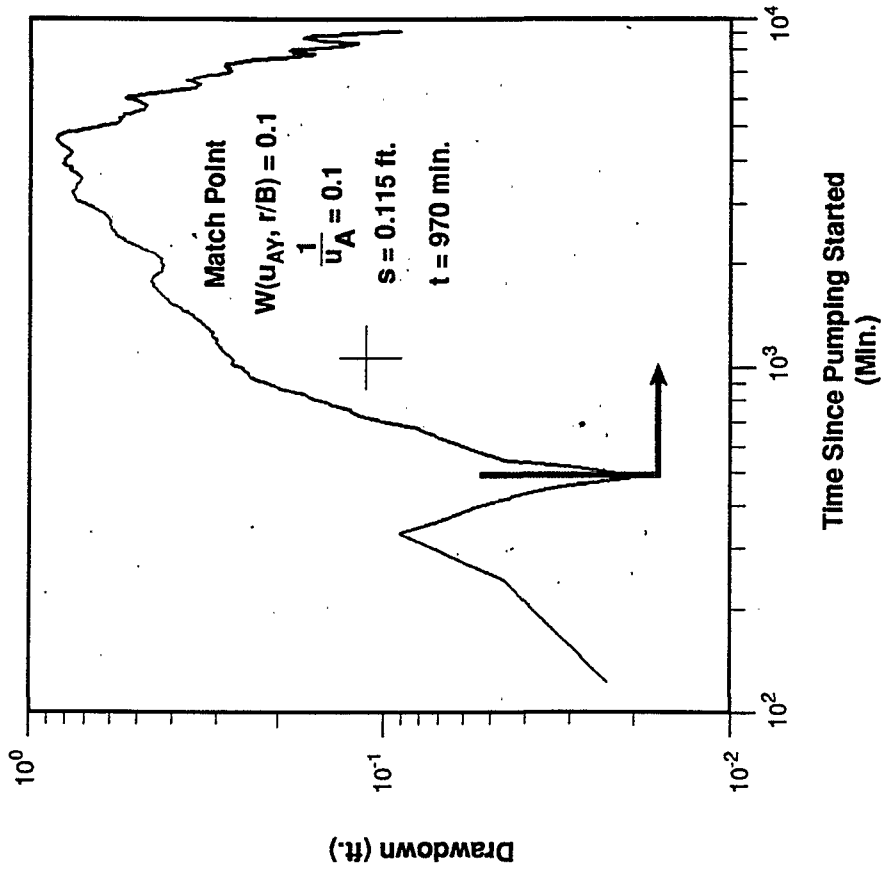
T	=	transmissivity (ft <sup>2</sup> /day)
Q	=	discharge = 13 gpm = 2,503 cubic feet per day
W (u <sub>AY</sub> , r/B)	=	0.1
s	=	drawdown = 0.115 feet
t	=	970 minutes = 0.67 days
u <sub>A</sub>	=	1.0
r	=	distance between wells = 1,100 ft

The results were

T	=	173 ft <sup>2</sup> /day
S	=	0.00038

This transmissivity is somewhat lower than the values obtained from the pumping test in well S-9 and probably reflects the lower yield of the GKM well.

Barometric pressure measurements were collected at the GKM well through the pumping test, and additional data were obtained from the Hatfield airport. The barometric data were plotted and compared with the pumping test data. No trends or cycles were noted in the barometric data, and there was no correlation to the drawdown observed in NPWA Well S-9.



AR300726

**Figure 3-10**  
DRAWDOWN IN WELL S-9 CORRECTED FOR  
BACKGROUND DRAWDOWN  
North Penn Area 1  
Phase II RI/FS



The data from the pumping test indicate that the radius of influence achieved during pumping of the GKM well extended to well S-9. The radius of influence of the GKM well in other directions cannot be determined because of the lack of other observation wells in the aquifer. The aquifer, consisting of fractured bedrock, will probably not have an isotropic cone of influence.

As described by Longwill and Wood (1965), wells installed in the fractured bedrock in the North Penn area at locations across the bedrock strike (approximately northwest and southeast) from a pumping well may not be open to the same interval as the pumping well (assuming the same well depth) due to the northwest dip of the sedimentary rock beds. On the other hand, wells located along the bedrock strike (approximately northeast and southwest) may be open to the same interval as the pumping well (assuming the same well depth). As a result, little or no drawdown may be observed to the northwest and southeast because of a pumping well, whereas considerable drawdown may be observed to the northeast and southwest. The result is an elliptical cone of depression oriented approximately northeast-southwest.

WDCR657/014.51



**Section 4**  
**Nature and Extent of Contamination**

AR300728

## Section 4 Nature and Extent of Contamination

### 4.1 Introduction

During the remedial investigation (RI) activities at the site, the nature and extent of soil and groundwater contamination by volatile organic compounds (VOCs) were investigated. The results of the investigation are presented in this section. The data collected during the RI field activities was validated in the manner described below.

The EPA Sample Management Office (SMO) received data packages from the laboratories in the CLP and distributed them to the Contract Project Management Section (CPMS) of the CRL. The CPMS reviews all data packages generated from regional sampling efforts.

After receipt of the CRL-reviewed data packages, the data were validated and summarized by CH2M HILL in accordance with *Region III General Guidance for Data Review*, June 1992. During validation, any data that should be qualified were flagged with the appropriate symbol. Results for field blanks and field duplicates were reviewed and the data further qualified if necessary. Finally, the data set as a whole was examined for consistency, anomalous results, and reasonableness. A data validation report was submitted to the CRL and the RPM.

### 4.2 Soils

The nature and extent of VOC contamination in soils at each of the five facilities were investigated to determine if any of the facilities may have been sources of contamination to the groundwater and whether any of the facilities may continue to be sources. Soil samples were collected at 2-foot intervals and screened with an HNU photoionization detector (PID). Selected soil samples were analyzed in the field with a portable gas chromatograph (GC) in a close support laboratory (CSL). The procedures used in the CSL are described in Section 2 and in Appendix C. Selected soil samples were sent through the EPA Contract Laboratory Program (CLP). The soil samples sent through the CLP program were selected to provide information on the areal extent of contamination at the facility and to quantify levels of contamination at the facility for use in the risk assessment.

Soil samples were collected at 2-foot intervals in each of the borings to auger refusal. Samples were logged and collected from each interval. Samples were screened with a photoionization detector for the presence of volatile organics. At each facility, six samples were planned for CLP analysis. To determine which samples would be analyzed through the CLP, the field screening and field GC data were used to identify the extent of contamination. Samples sent through the CLP were selected to confirm

the magnitude and scope of contamination at each facility. Soil samples were selected for CLP and field GC analysis on the basis of the appearance and odor of the soil. Samples were selected for field GC analysis to provide areal and vertical delineation of soil contamination. Samples were composited for nonvolatiles analyses to meet volume requirements.

The results of the field GC analyses of the soil samples are presented in Table 4-1. Validated results of the CLP analyses of selected samples are summarized in Table 4-2. The complete results of CLP analyses are provided in Appendix F.

Acetone and methylene chloride were detected in most of the samples sent to the CLP. The presence of these VOCs was the result of contamination in the analytical procedures during analysis as demonstrated by detection of these compounds in the associated method blanks. All of the soil samples sent to the CLP contained one or both of these chemicals, even those samples from facilities where no other contaminants were detected.

#### 4.2.1 Gentle Cleaners

At Gentle Cleaners, soil samples were collected from eight soil borings. Eleven soil samples were analyzed with the field GC for VOCs. Five of these samples and one duplicate were then sent through the CLP for analysis for VOCs and total organic compounds (TOCs).

The chemical analyses indicated that PCE is the primary contaminant at this facility. The soil boring locations and field GC results for PCE are shown in Figure 4-1, and the PCE results of the CLP-analyzed samples are presented in Figure 4-2. The highest concentrations of PCE were detected in samples from the 6- to 10-foot interval in the backyard of Gentle Cleaners. One of these samples was sent to the CLP and contained a PCE concentration of 300,000  $\mu\text{g}/\text{kg}$ . This sample exceeded the detection limits of the field GC, as did a sample from boring GC-7; this exceedance is indicated by the term "off scale" in the figure. The highest concentration of PCE in a soil sample obtained from the yard next to the cleaners was from boring GC3, the nearest boring to the stone wall separating the two properties. Concentrations were low or below detection limits in surface soils at this facility.

TCE was detected by the field GC at 1  $\mu\text{g}/\text{kg}$  in one sample—in the 8- to 10-foot interval at boring GC6—but none was detected in samples sent to the CLP. No other VOCs were detected in soil samples by the field GC.

A leaf sample was collected from the GC-7 location when it was noticed that the leaves had a strong solvent smell, and VOCs registered strongly on the HNU. The sample was analyzed using the field GC. No PCE was detected in the sample but several compounds not identifiable by the field GC were present.

**TABLE 4-1  
NORTH PENN AREA 1 SOIL ANALYSIS  
FIELD GC RESULTS**

Sample Identification	Depth (feet)	Detected Compounds	Concentrations ( $\mu\text{g}/\text{kg}$ )
<b>LEXCO</b>			
L4-S1	0-2	ND	
L5-S2	2-4	ND	
L5-S3	4-6	ND	
L4-S3	4-6	ND	
L1-S1	0-2	ND	
L6-S2	2-4	ND	
L6-S4	6-8	ND	
L3-S1	0-2	ND	
L2-S1	0-2	ND	
L8-S1	0-2	ND	
L8-S2	2-4	ND	
L9-S2	2-4	ND	
L7-S2	2-4	ND	
<b>GRANITE KNITTING MILLS</b>			
GKM2-S2	2-4	Tetrachloroethylene	19
GKM2-S5	8-10	Tetrachloroethylene	152
GKM1-S4	6-8	ND	
GKM1-S3	4-6	Tetrachloroethylene	4
GKM1-S1	0-2	Tetrachloroethylene	3
GKM6-S4	6-8	Tetrachloroethylene	45
GKM6-S4D	6-8	Tetrachloroethylene	37
GKM5-S5	8-10	Tetrachloroethylene	4
GKM6-S6	10-12	Tetrachloroethylene	4
GKM5-S3	4-6	Tetrachloroethylene	9

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**TABLE 4-1  
NORTH PENN AREA 1 SOIL ANALYSIS  
FIELD GC RESULTS**

<b>Sample Identification</b>	<b>Depth (feet)</b>	<b>Detected Compounds</b>	<b>Concentrations (µg/kg)</b>
GKM5-S4	6-8	Tetrachloroethylene	10
GKM4-S3	4-6	cis-1,2-Dichloroethylene	3.2
		Trichloroethylene	2.2
		Tetrachloroethylene	52
GKM7-S3	4-6	Tetrachloroethylene	7
GKM10-S2	2-4	Tetrachloroethylene	15
GKM3-S2	2-4	Tetrachloroethylene	6
GKM3-S4	6-8	ND	
GKM8-S4	6-8	Trans-1,2-Dichloroethylene	175
		cis-1,2-Dichloroethylene	7
		Trichloroethylene	53
		Tetrachloroethylene	off-scale
GKM14-S1	0-2	ND	
GKM14-S2	2-4	ND	
GKM11-S1	0-2	ND	
GKM13-S1	0-2	Trichloroethylene	4
		Tetrachloroethylene	582
GKM12-S2	2-4	Tetrachloroethylene	2
GKM11-S2	2-4	Tetrachloroethylene	1
GKM12-S1	0-2	ND	
GKM Storm Drain	0-1	ND	
<b>STANDARD TERRY</b>			
ST3-S3	4-6	ND	
ST3-S5	8-10	ND	
ST2-S5	8-10	ND	

AR300732

**TABLE 4-1  
NORTH PENN AREA 1 SOIL ANALYSIS  
FIELD GC RESULTS**

<b>Sample Identification</b>	<b>Depth (feet)</b>	<b>Detected Compounds</b>	<b>Concentrations (µg/kg)</b>
ST5-S2	2-4	ND	
ST8-S3	4-6	ND	
ST4-S8	14-16	ND	
ST9-S5	8-10	ND	
ST5-S4	6-8	ND	
ST6-S4	68	ND	
ST1-S3	4-6	ND	
ST3-S7	12-14	ND	
ST7-S2	2-4	ND	
ST10-S4	6-8	ND	
ST5-S1	0-2	ND	
ST9-S3	4-6	ND	
ST4-S5	8-10	ND	
ST4-S7	12-14	ND	
<b>GENTLE CLEANERS</b>			
GC5-S1	0-2	ND	
GC3-S3	4-6	Tetrachloroethylene	1449
GC7-S2	2-4	ND	
GC7-S4	6-8	Tetrachloroethylene	off-scale
GC1-S2	2-4	Tetrachloroethylene	2
GC6-S5	8-10	Trichloroethylene	1
		Tetrachloroethylene	off-scale
GC4-S1	0-2	ND	
GC8-S3	4-6	Tetrachloroethylene	73
GC2-S1	0-2	ND	

**TABLE 4-1  
NORTH PENN AREA 1 SOIL ANALYSIS  
FIELD GC RESULTS**

Sample Identification	Depth (feet)	Detected Compounds	Concentrations (µg/kg)
GC6-S6	10-12	Tetrachloroethylene	28
GC8-S2	2-4	Tetrachloroethylene	53
GC-TS	0-0.5	Tetrachloroethylene	5
<b>PARKSIDE APARTMENTS</b>			
PA2-S2	2-4	Trans-1,2-Dichloroethylene	0
		cis-1,2-Dichloroethylene	16
		Trichloroethylene	50
		Tetrachloroethylene	193
PA3-S2	2-4	Tetrachloroethylene	3
PA8-S4	6-8	Tetrachloroethylene	5
PA4-S3	4-6	cis-1,2-Dichloroethylene	7
		Trichloroethylene	5
		Tetrachloroethylene	160
PA5-S3	4-6	Tetrachloroethylene	21
PA7-S6	10-12	Tetrachloroethylene	23
PA6-S4	6-8	Trichloroethylene	4
		Tetrachloroethylene	1787
PA7-S3	4-6	Tetrachloroethylene	5
PA10-S2	2-4	ND	
PA9-S3	4-6	Tetrachloroethylene	17
PA5-S2	2-4	Tetrachloroethylene	32
PA8-S1	0-2	Tetrachloroethylene	3

NOTE: ND = None detected.

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**Table 4-2  
ANALYSES OF ORGANIC COMPOUNDS IN SOILS  
CLP RESULTS**

Gentle Cleaners								
Sample No.	GC-3	GC-5	GC-6	GC-7	GC-7 (DUP)	GC-8		
Depth (ft)	4-6	0-2	8-10	2-4	2-4	4-6		
Compound								
1,2-DCA ( $\mu\text{g}/\text{kg}$ )	<13	<12	<11	<11	<12	<12		
MEK ( $\mu\text{g}/\text{kg}$ )	<13	<12	<11	<11	<12	<12		
Acetone ( $\mu\text{g}/\text{kg}$ )	43 B	12 B	27 B	7 B	14 B	48 B		
MC ( $\mu\text{g}/\text{kg}$ )	17 B	18 B	15 B	14 B	12 B	18 B		
PCE ( $\mu\text{g}/\text{kg}$ )	100	23 J	300,000	<11	<12	190		
Toluene ( $\mu\text{g}/\text{kg}$ )	<13	<12	<11	<11	<12	<12		
TCE ( $\mu\text{g}/\text{kg}$ )	<13	<12	<11	<11	<12	<12		
TOC (mg/kg)	1,490	2,450	21,800	20,800	25,000	1,680		
Granite Knitting Mills								
Sample No.	GKM-3	GKM-4	GKM-5	GKM-7	GKM-8	GKM-10	GKM-12	GKM-13
Depth (ft)	2-4	4-6	6-8	4-6	6-8	2-4	2-4	0-2
Compound								
1,2-DCA ( $\mu\text{g}/\text{kg}$ )	<12	<11	<12	<12	45	<12	<11	<12
MEK ( $\mu\text{g}/\text{kg}$ )	17 B	<11	<12	<12	<12	<12	<11	<12
Acetone ( $\mu\text{g}/\text{kg}$ )	25 B	24 B	22 B	31 B	21 B	13 B	12 B	20 B
MC ( $\mu\text{g}/\text{kg}$ )	12 B	11 B	11 B	12 B	12 B	14 B	33 B	53 B
PCE ( $\mu\text{g}/\text{kg}$ )	<12	27	6 J	8 J	6,900	6 J	<11	1,000
Toluene ( $\mu\text{g}/\text{kg}$ )	<12	3 J	<12	<12	<12	12 L	<11	<12
TCE ( $\mu\text{g}/\text{kg}$ )	<12	<11	<12	<12	64	<12	<11	<12
TOC (mg/kg)	766	4,640	548	2,660	5,320	3,580	1,340	25,000



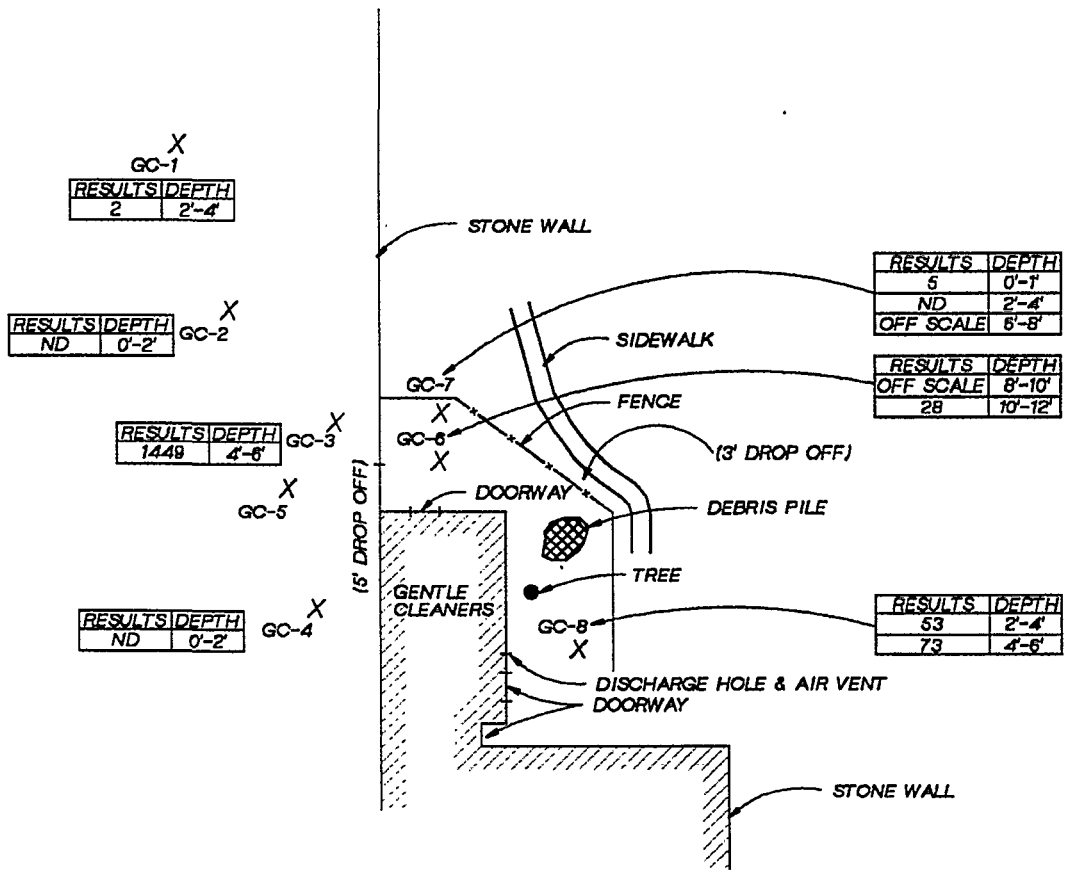
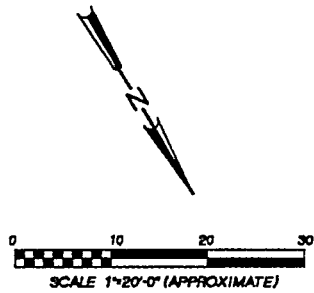
**Table 4-2  
ANALYSES OF ORGANIC COMPOUNDS IN SOILS  
CLP RESULTS**

Parkside Apartments							
Sample No.	PA-2	PA-3	PA-5	PA-7	PA-8	PA-10	PA-10 (DUP)
Depth (ft)	2-4	2-4	4-6	10-12	6-8	2-4	2-4
Compound							
1,2-DCA ( $\mu\text{g}/\text{kg}$ )	4 J	<12	<12	<11	<12	<12	<12
MEK ( $\mu\text{g}/\text{kg}$ )	<12	<12	<12	<11	<12	<12	<12
Acetone ( $\mu\text{g}/\text{kg}$ )	<12	8 B	9 B	15 B	18 B	12 B	11 B
MC ( $\mu\text{g}/\text{kg}$ )	48 B	38 B	34 B	53 B	17 B	13 B	15 B
PCE ( $\mu\text{g}/\text{kg}$ )	99	<12	5 J	120 J	56	<12	<12
Toluene ( $\mu\text{g}/\text{kg}$ )	<12	<12	<12	<11	<12	<12	<12
TCE ( $\mu\text{g}/\text{kg}$ )	29	<12	<12	<11	<12	<12	<12
TOC (mg/kg)	4,520	2,540	6,050	15,800	7,990	4,160	4,530
Lexco Engineering							
Sample No.	L-2	L-4	L-6	L-7	L-8	L-9	
Depth (ft)	0-2	0-2	2-4	2-4	2-4	2-4	
Compound							
1,2-DCA ( $\mu\text{g}/\text{kg}$ )	<12	<12	<11	<12	<12	<12	
MEK ( $\mu\text{g}/\text{kg}$ )	<12	<12	<11	<10	<12	<12	
Acetone ( $\mu\text{g}/\text{kg}$ )	22 B	22 B	20 B	20 B	21 B	52 B	
MC ( $\mu\text{g}/\text{kg}$ )	45 B	44 B	36 B	41 B	43 B	45 B	
PCE ( $\mu\text{g}/\text{kg}$ )	<12	<12	<11	<12	<12	<12	
Toluene ( $\mu\text{g}/\text{kg}$ )	<12	<12	<11	<12	<12	<12	
TCE ( $\mu\text{g}/\text{kg}$ )	<12	<12	<11	<12	<12	<12	
TOC (mg/kg)	1,190	2,010	458	3,600	227	455	

**Table 4-2  
ANALYSES OF ORGANIC COMPOUNDS IN SOILS  
CLP RESULTS**

Standard Terry Mills						
Sample No.	ST-2	ST-3	ST-5	ST-8	ST-9	ST-10
Depth (ft)	8-10	12-14	4-6	4-6	8-10	6-8
Compound						
1,2-DCA ( $\mu\text{g}/\text{kg}$ )	<12	<11	<12	<12	<12	<13
MEK ( $\mu\text{g}/\text{kg}$ )	<12	<11	<12	<12	<12	<13
Acetone ( $\mu\text{g}/\text{kg}$ )	70 B	25 B	41 B	10 B	81 B	17 B
MC ( $\mu\text{g}/\text{kg}$ )	14 B	9 B	14 B	21 B	110 B	29 B
PCE ( $\mu\text{g}/\text{kg}$ )	<12	<11	<12	<12	7 J	<13
Toluene ( $\mu\text{g}/\text{kg}$ )	<12	<11	<12	<12	<12	<13
TCE ( $\mu\text{g}/\text{kg}$ )	<12	<11	<12	<12	<12	<13
TOC (mg/kg)	2,433	20,700	25,000	25,000	630	6,990
<p>Notes: DCA = Dichloroethane                      DUP = Duplicate sample            MEK = Methyl ethyl ketone                    TCE = Trichloroethene            MC = Methylene chloride                      TOC = Total organic carbon            PCE = Tetrachloroethene            J indicates that the sample contains the compound at an estimated concentration.            B indicates that the compound was detected in the associated method blank.</p>						

WDCR675/011.51



**LEGEND**

- X SOIL BORING LOCATIONS
- ND NOT DETECTED
- ALL UNITS IN  $\mu\text{g}/\text{kg}$

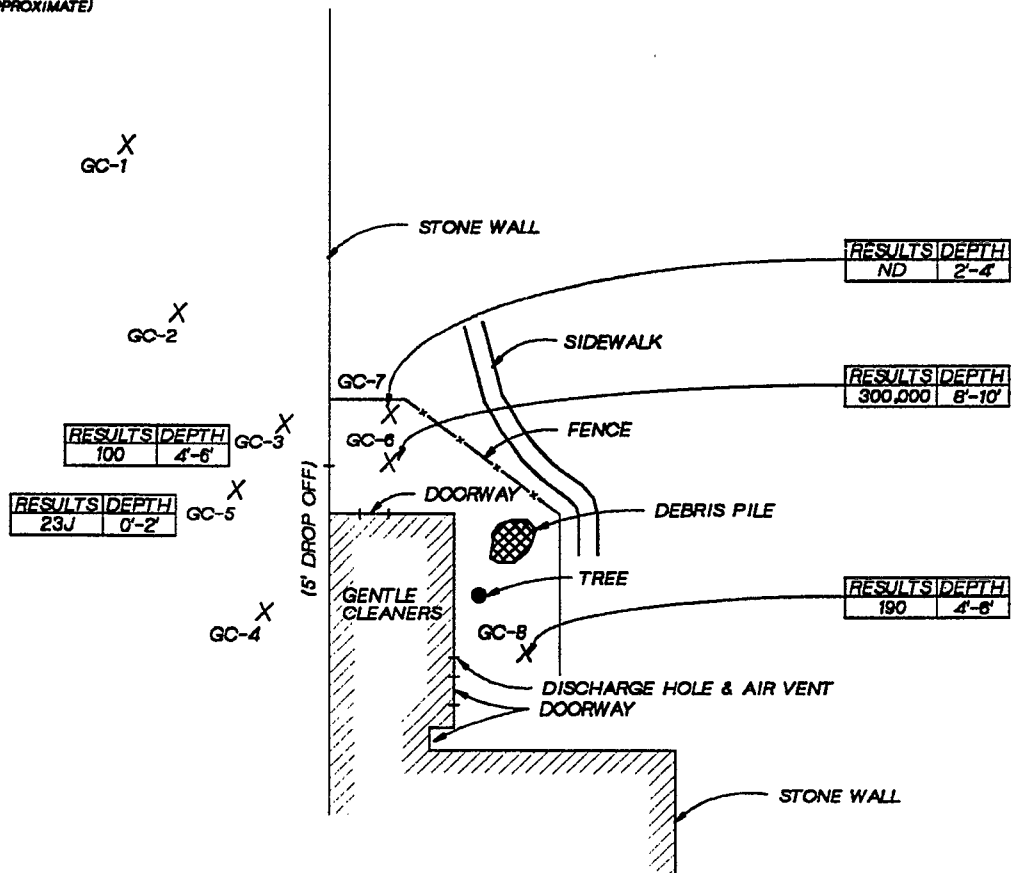
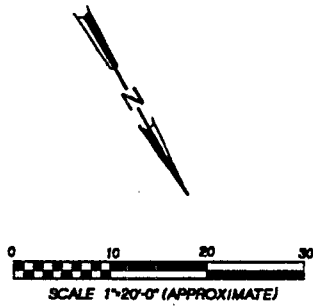
**FIGURE 4-1**

PCE CONCENTRATIONS  
FROM FIELD GC ANALYSES  
GENTLE CLEANERS

North Penn Area 1  
Phase II RI/FS



AR300738



**LEGEND**

- X SOIL BORING LOCATIONS
- ND NOT DETECTED
- ALL UNITS IN  $\mu\text{g}/\text{kg}$
- J VALUE ESTIMATED BELOW DETECTION LIMIT

**FIGURE 4-2**

**PCE CONCENTRATIONS FROM CLP ANALYSES GENTLE CLEANERS**  
 North Penn Area 1  
 Phase II RI/FS



AR300739

Contaminant levels determined during this investigation suggest that the extent of soil contamination at this facility is limited. The highest levels were found behind the cleaners, and some elevated levels were found in the adjoining yard. But no VOCs were detected in soil samples taken south of the cleaners, at borings GC1 and GC2, along a probable path of contaminant migration, down the gentle slope of the yard. Access limitations prevented obtaining samples southwest and west of the cleaners.

The concentration of 300,000  $\mu\text{g}/\text{kg}$  of PCE detected in one sample at this facility suggests the presence, at least at some time in the past, of dense non-aqueous phase liquids (DNAPLs). Some 75 gallons of PCE reportedly were spilled at this facility. This liquid would have seeped down into the soil and would have been adsorbed to some degree onto the soil particles.

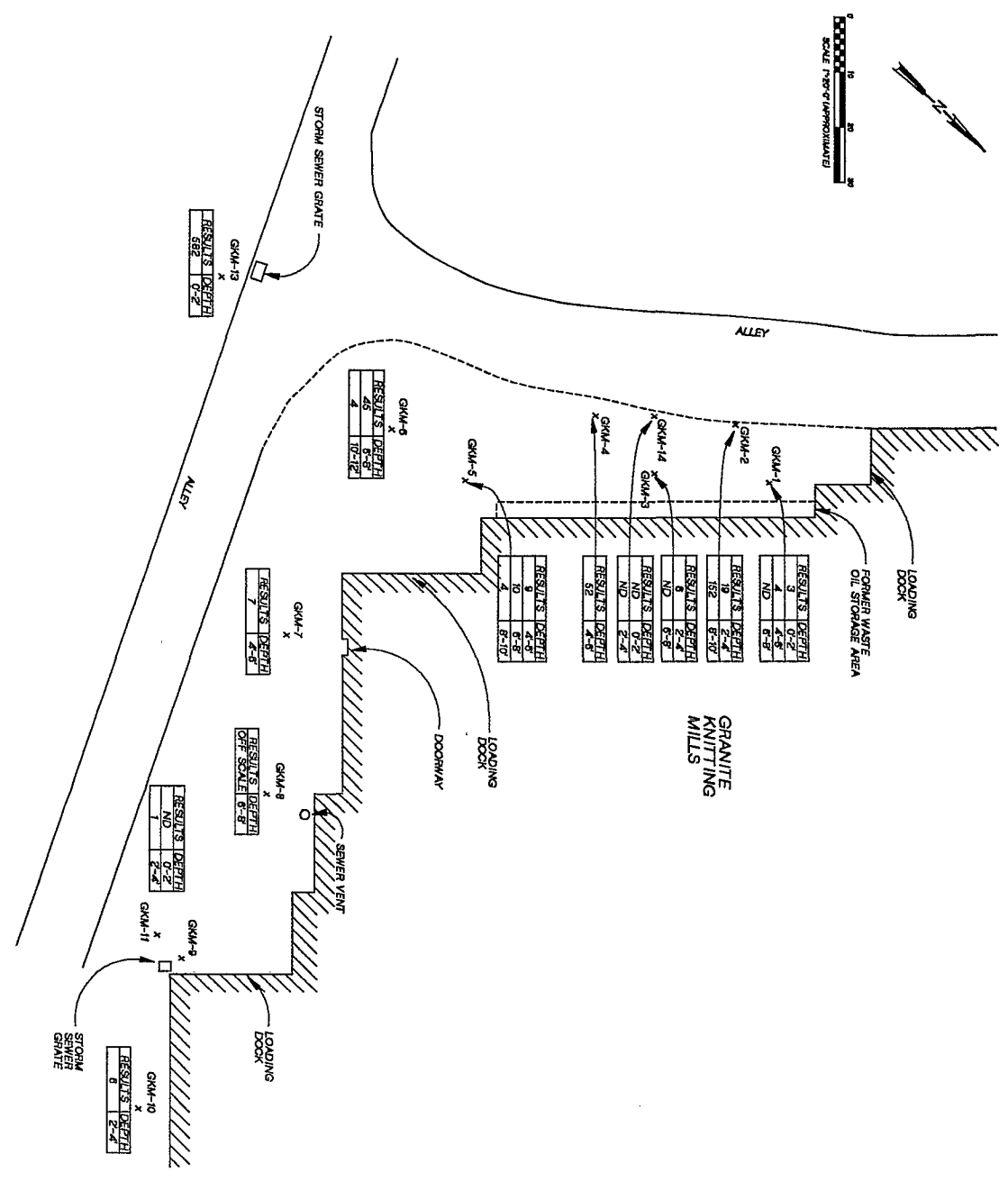
#### **4.2.2 Granite Knitting Mills**

The soils at Granite Knitting Mills were investigated by drilling and sampling 14 soil borings. All of the soil borings were drilled through asphalt cover and the underlying soils to the top of bedrock. Twenty-four soil samples were analyzed with the field GC, and eight samples were sent to the CLP.

The results of the field GC data indicate that the contaminants in the soils associated with this facility were primarily PCE. The soil boring locations and field GC sampling results for PCE are presented in Figure 4-3. Selected samples were sent through the EPA CLP program for analysis and documentation. The PCE results of the CLP sampling are presented in Figure 4-4. Boring GKM-12 is not shown in either Figure 4-1 or 4-2. Boring GKM-12 was located in the alley at the northeast corner of the mills property (off of the map on the right-hand side) to help determine if any contamination was moving down the hill from Gentle Cleaners and running into the alley.

Soil samples were collected to auger refusal at 4 feet below the surface. The 2-to-4 foot sample was analyzed by the field GC and through the CLP. The field GC detected PCE at 2  $\mu\text{g}/\text{kg}$ . The CLP sample had a detection limit of 11  $\mu\text{g}/\text{kg}$  and did not detect PCE in the sample.

The highest concentration of detected PCE at the facility was at boring GKM-8; 6,900  $\mu\text{g}/\text{kg}$  were detected by the CLP, and the sample exceeded the field GC detection limit. Most other concentrations were low, except at boring GKM-13, where 1,000  $\mu\text{g}/\text{kg}$  of PCE were detected in the surface sample (below asphalt) by the CLP. This location is near a storm sewer grate, suggesting the possibility that contamination may move into the storm sewer with surface runoff. No VOCs were detected in a sample of soil obtained from the northern storm drain and analyzed by the field GC (sample GKM storm drain).



**LEGEND**  
 X SOIL BORING LOCATIONS  
 ND NOT DETECTED  
 ALL UNITS IN µg/mg  
 J VALUE ESTIMATED BELOW DETECTION LIMIT

**FIGURE 4-3**  
 PCE CONCENTRATIONS FROM FIELD GC ANALYSES GRANITE KNITTING MILLS North Park Area Phase II RI/FS

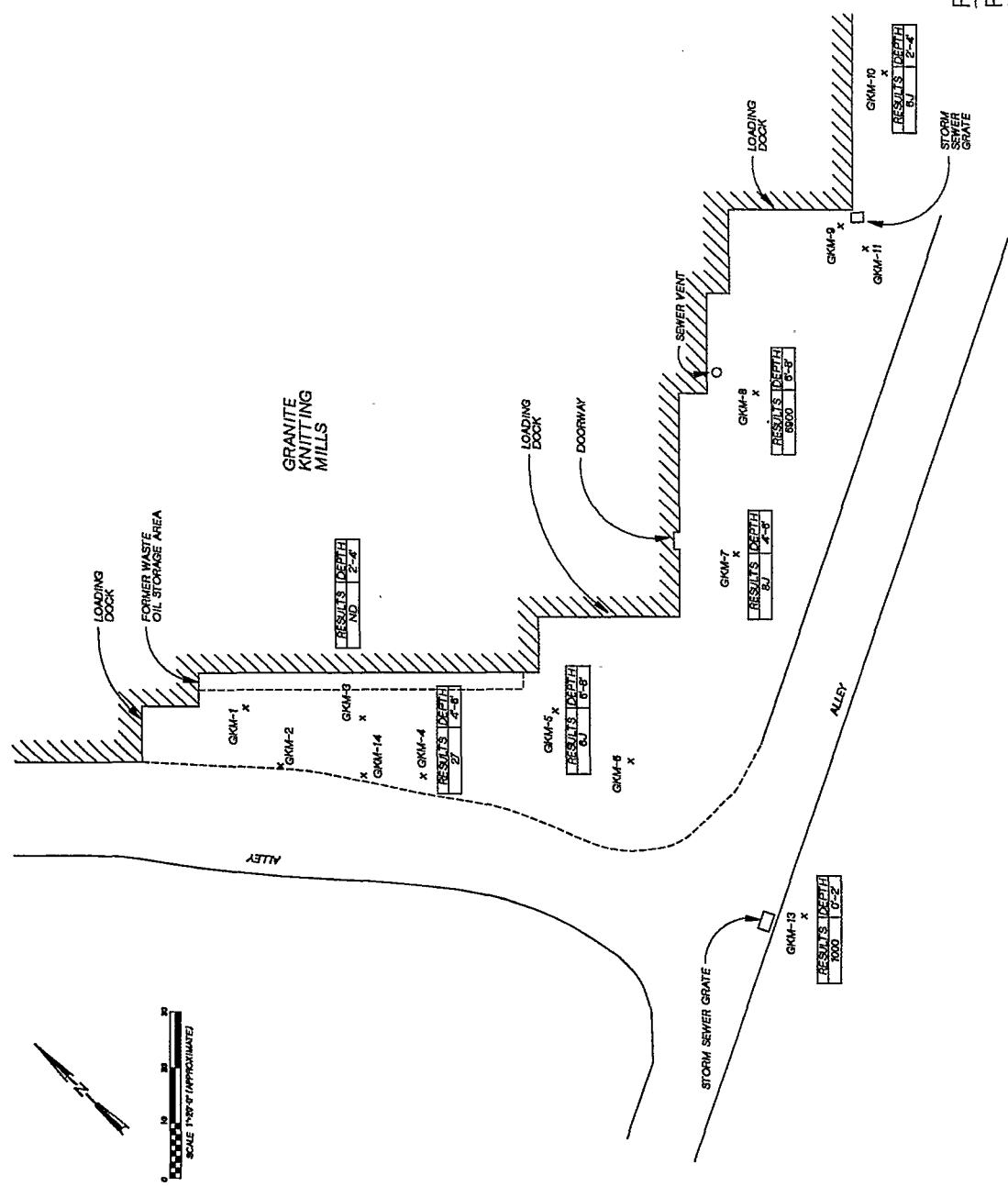
AP 300741



AR300741

**FIGURE 4-4**  
**PCE CONCENTRATIONS**  
**FROM CLP ANALYSES**  
**GRANITE KNITTING MILLS**  
 North Penn Area 1  
 Phase II RI/FS

**LEGEND**  
 x SOIL BORING LOCATIONS  
 NO NOT DETECTED  
 ALL UNITS IN  $\mu\text{g}/\text{kg}$   
 J VALUE ESTIMATED BELOW DETECTION LIMIT



Surface water samples were collected from the unnamed tributary to Skippack Creek below the outfall of this storm sewer when it was discharging. The surface water samples were analyzed with the field GC, and no VOCs were detected.

The soil samples collected from the former solvent storage area (along the southwest side of the building) found PCE levels varying from not detected to 152  $\mu\text{g}/\text{kg}$ . The PCE levels found in boring GKM-8 were significantly higher.

Other VOCs were detected by the field GC and the CLP: cis-1,2-DCE (two detections), trans-1,2-DCE (one detection), 1,2-DCA (one detection), methyl ethyl ketone (one detection), toluene (two detections), and TCE (three detections). The highest levels of most of these other VOCs were detected by the field GC in the 6- to 8-foot sample from GKM-8—trans-1,2-DCE at 175  $\mu\text{g}/\text{kg}$ , cis-1,2-DCE at 7  $\mu\text{g}/\text{kg}$ , and TCE at 53  $\mu\text{g}/\text{kg}$ ; 1,2-DCA also was detected at 45  $\mu\text{g}/\text{kg}$  by the CLP. The two other field-GC detections of other VOCs were below 5  $\mu\text{g}/\text{kg}$ . Acetone and methylene chloride also were reported as being detected by the CLP in all of the samples.

The results of soil analyses for VOCs at this facility suggest that the highest levels of contamination are restricted to the southeast side of the building. The highest level of contamination in this area was found at the interval from 6 to 8 feet. Concentrations at the surface were low to nondetectable.

#### 4.2.3 Parkside Apartments

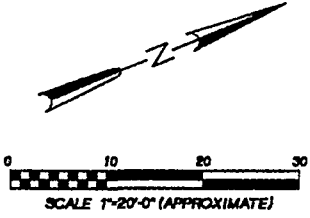
The Parkside Apartments facility was investigated through the drilling and sampling of 10 soil borings in the lawn behind the building. Twelve selected soil samples were analyzed in the field using a field GC. Six samples and one duplicate were then sent through the CLP. On the basis of the known past property use, the soil borings were located to provide coverage of the area of the lawn where contamination may have occurred.

The analyses of the soil samples found contamination to be primarily PCE. The results of the field GC analyses for PCE are presented in Figure 4-5. The results of the CLP analyses for PCE are presented in Figure 4-6. The levels of PCE observed ranged from not detected to 1,787  $\mu\text{g}/\text{kg}$  on the field GC, and not detected to 120  $\mu\text{g}/\text{kg}$  in the CLP data. The highest levels of PCE were located next to the building at borings PA-2, PA-4, and PA-6. Away from the building, the levels of PCE were much lower.

Cis-1,2-DCE (two detections), trans-1,2-DCE (one detection), TCE (three detections), and 1,2-DCA (one detection) also were detected at the facility. All of these other VOC detections were in samples obtained from borings PA-2, PA-4, and PA-6, next to the building. The highest concentration of TCE detected was 50  $\mu\text{g}/\text{kg}$  in a 2- to 4-foot sample in boring PA-2. Acetone and methylene chloride also were reported as being detected by the CLP in most of the samples.



UNNAMED TRIBUTARY OF SKIPPACK CREEK



WOODS

PA-8  
X

RESULTS	DEPTH
3	0'-2'
5	6'-8'

PA-7  
X

RESULTS	DEPTH
5	4'-6'
23	10'-12'

PA-9  
X

RESULTS	DEPTH
17	4'-6'

PA-6  
X

RESULTS	DEPTH
1787	6'-8'

PA-10  
X

RESULTS	DEPTH
ND	2'-4'

PARKSIDE APARTMENTS

PA-6  
X

RESULTS	DEPTH
32	2'-4'
21	4'-6'

PA-4  
X

RESULTS	DEPTH
180	4'-6'

PA-3  
X

RESULTS	DEPTH
3	2'-4'

PA-2  
X

RESULTS	DEPTH
193	2'-4'

FORMER DRY CLEANER

PA-1  
X

WOODS

STONE WALL

**LEGEND**

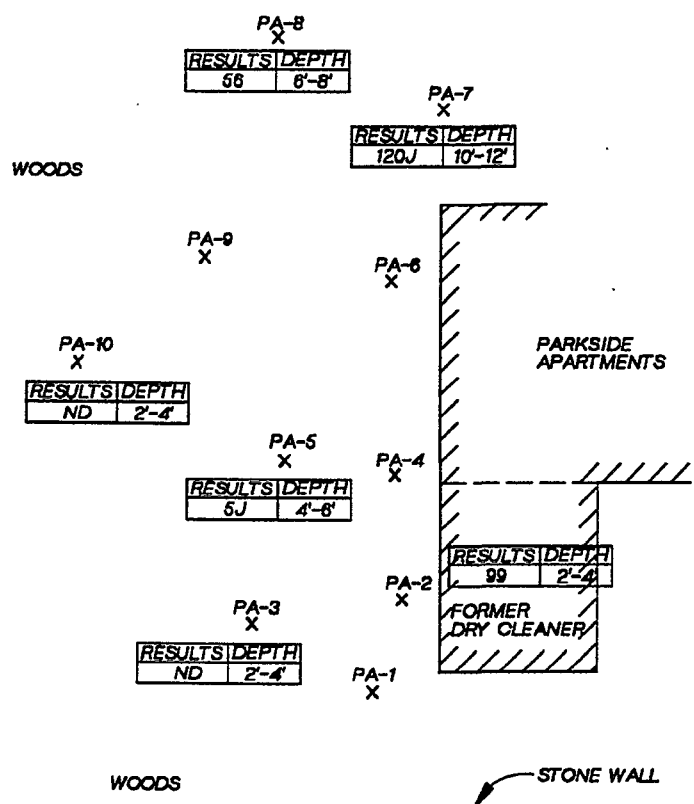
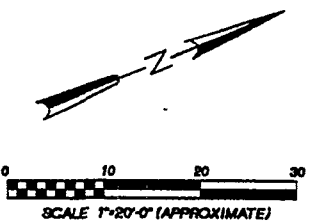
- X SOIL BORING LOCATIONS
- ND NOT DETECTED
- ALL UNITS IN µg/kg

**FIGURE 4-5**  
PCE CONCENTRATIONS  
FROM FIELD GC ANALYSES  
PARKSIDE APARTMENTS  
North Penn Area 1  
Phase II RI/FS



AR300744

UNNAMED TRIBUTARY OF SKIPPACK CREEK



**LEGEND**

- X SOIL BORING LOCATIONS
- ND NOT DETECTED
- ALL UNITS IN  $\mu\text{g}/\text{kg}$
- J VALUE ESTIMATED BELOW DETECTION LIMIT

**FIGURE 4-6**  
**PCE CONCENTRATIONS**  
**FROM CLP ANALYSES**  
**PARKSIDE APARTMENTS**  
 North Penn Area 1  
 Phase II RI/FS



AR300745

The results of soil analyses for VOCs at this facility suggest that the highest levels of contamination are restricted to the southwest side of the building. The highest level of contamination in this area was found at the interval from 6 to 8 feet. Concentrations at the surface were low.

#### **4.2.4 Lexco Engineering**

Nine soil borings were drilled and sampled at this location. Soil samples were collected and screened with an HNU for volatile organics. Twelve selected soil samples also were analyzed by a field GC for VOCs. Six samples were sent through the CLP laboratories for analysis.

The samples collected at Lexco Engineering were split by the PRP. The results of the split samples have not been provided to CH2M HILL.

The soil borings were located to provide areal coverage of the parking lot behind the facility. Soil samples were not collected adjacent to the heating oil UST because of overhead power lines.

No VOCs were detected in any of the soil samples analyzed.

#### **4.2.5 Standard Terry Mills**

Ten soil borings were drilled and sampled to define the presence and levels of soil contamination at this facility. The soil borings were installed in the area of the former loading dock and around the UST pad. The soil samples were collected from below the asphalt that covered most of this facility. Seventeen selected soil samples were analyzed for volatiles with a field GC, and six of those samples also were analyzed for VOCs through the CLP laboratories. No VOCs were detected by the field GC in any of the soil samples from this facility. PCE, at an estimated concentration of 7  $\mu\text{g}/\text{kg}$ , was detected in one CLP sample.

#### **4.2.6 Summary**

Detectable levels of VOCs (particularly PCE) in soil were identified at Gentle Cleaners, Granite Knitting Mills, and Parkside Apartments. PCE, at an estimated concentration of 7  $\mu\text{g}/\text{kg}$ , was found at Standard Terry Mills. No VOCs were detected at Lexco. A concentration of PCE of 300,000  $\mu\text{g}/\text{kg}$  detected at Gentle Cleaners suggests the probable presence of DNAPLs in the soils, at least at some time in the past; this is consistent with the reported spill of 75 gallons of PCE at the facility previously.

### 4.3 Groundwater

VOCs, particularly PCE and TCE, have been detected in groundwater at the site. The groundwater was sampled as part of this RI at several wells in and near the site. The wells sampled during this investigation met one or more of the following criteria:

- They were identified during the well inventory as drilled wells.
- They had contaminants detected in previous investigations at the site.
- They are currently used as drinking water supply wells.
- They are located within 1/2 mile of the site boundaries.
- The owner allowed them to be sampled.

The results of the groundwater sampling and CLP analyses are presented in Table 4-3 and in Figure 4-7. The complete results of CLP analyses are presented in Appendix F. The highest levels of contamination occur in the wells at the center of the site, including the GKM well, NPWA wells S-9 and S-10, and the Souderton Borough well (well 679), which had PCE concentrations ranging from less than 1 to 5  $\mu\text{g/l}$ . The only other bedrock well exhibiting detectable levels of PCE was residential well R-2, with a concentration of 0.1  $\mu\text{g/l}$ .

The greatest variety of contaminants was detected in the GKM well, followed by the well at the Mennonite Home for the Aged (well R-10). The fact that the well at the Home did not contain PCE and had some other differences in the make-up of the VOCs detected suggests that the Home well may not be contaminated by the same plume as the other wells at the site.

Other VOCs identified in groundwater from the 13 wells sampled were benzene (two detections), bromodichloromethane (two detections), carbon disulfide (three detections), chloroform (two detections), chloromethane (nine detections), ethylbenzene (eight detections), and toluene (five detections). These VOCs were distributed widely among the wells, without being limited to either municipal wells only or residential wells only. It should be noted that except for 2  $\mu\text{g/l}$  of chloromethane estimated in a sample from well R-4, the concentrations of all of these VOCs are less than or equal to 1  $\mu\text{g/l}$ .

It is expected that all of the contaminated groundwater migrating to the southwest down the valley from sources at the PRPs would be intercepted by pumping wells S-8 and S-10. However, it is still possible that the PCE detected in residential well R-2 is derived from the same source as that in the main contaminant plume, especially in view of the similarity of contaminants in well R-2 and well S-9.

Residential wells R-17 and R-19 are both dug wells and do not therefore characterize the bedrock groundwater quality. However, because of their proximity to Gentle Cleaners, both have probably been contaminated by the cleaners as a result of migration of contaminants in the soil.

**Table 4-3  
ANALYSES OF ORGANIC COMPOUNDS IN GROUNDWATER SAMPLES  
OBTAINED FROM RESIDENTIAL, INDUSTRIAL, AND MUNICIPAL WELLS**

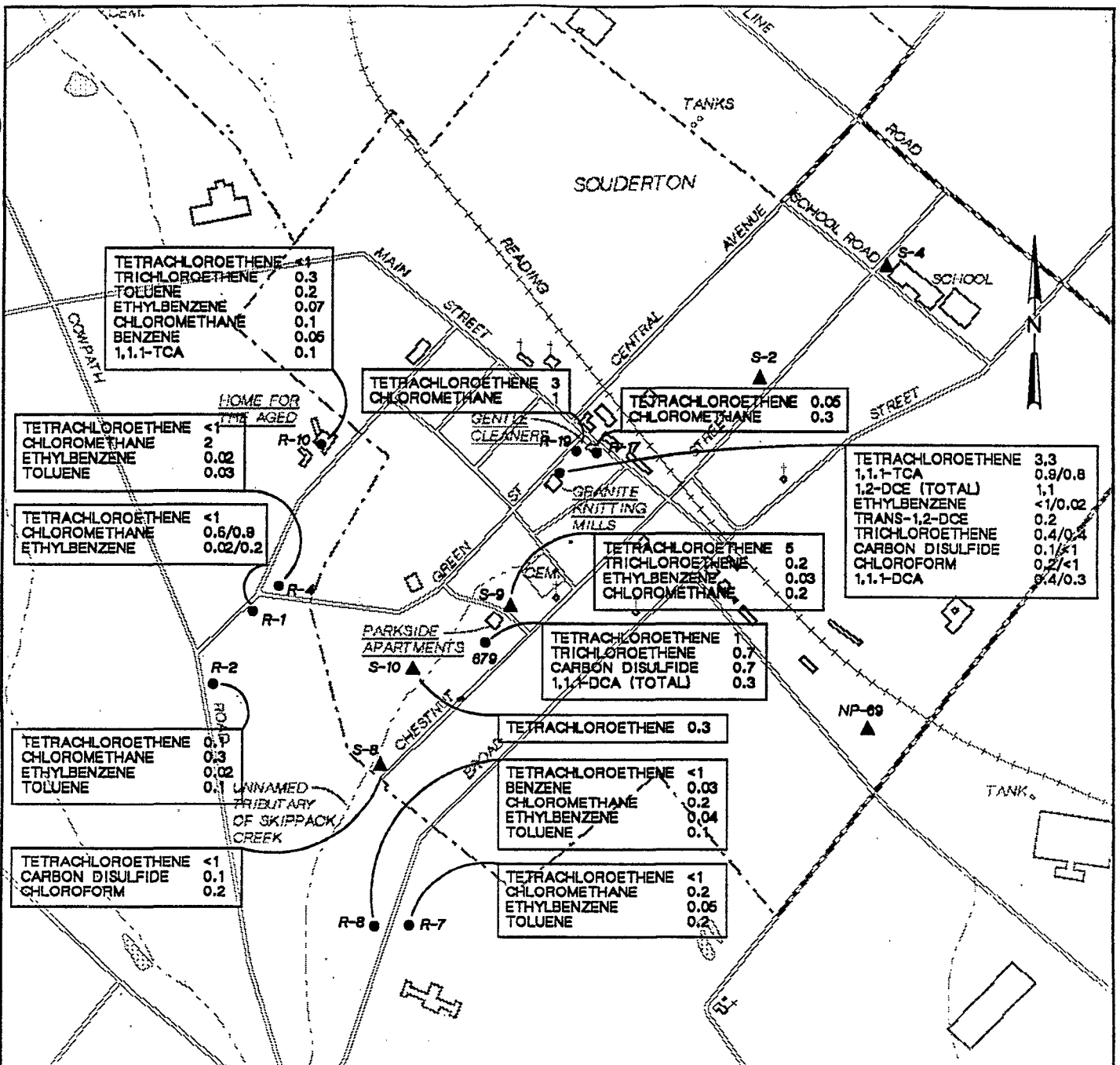
Well ID	R-1	R-1	R-2	R-4	R-7	R-8	Menonite Home R-10	R-17	R-19	NPWA S-8	NPWA S-9	NPWA S-10	GKM	Souderton Borough (679)	
Sample Number	R-1	GW1-02 (DUP)	GW6-01	GW2-01	GW8-01	GW7-01	GW9-01	GW4-01	GW3-01	S-8	GW5-01	S-10	GW10-01	GW11-01 (DUP)	GW12-01
<b>Compound</b>															
1,1,1-TCA (µg/l)	<1	<1	<1	<1	<1	<1	0.1 J	<1	<1	<1	<1	<1	0.9 J	0.8 J	<1
1,1-DCA (µg/l)	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	0.4 J	0.3 J	<1
Benzene (µg/l)	<1	<1	<1	<1	<1	0.03 J	0.05 J	<1	<1	<1	<1	<1	<1	<1	<1
BDCM (µg/l)	<1	<1	<1	<1	<1	<1	<1	<1	<1	0.1 J	<1	<1	0.1 J	0.1 J	<1
Carbon disulfide (µg/l)	<1	<1	<1	<1	<1	<1	<1	<1	<1	0.1 J	<1	<1	0.1 J	<1	0.7 J
Chloroform (µg/l)	<1	<1	<1	<1	<1	<1	<1	<1	<1	0.2 J	<1	<1	<1	0.2 J	<1
Chloromethane (µg/l)	0.6 J	0.9 J	0.3 J	2 J	0.2 J	0.2 J	0.1 J	0.3 J	1 J	<1	0.2 J	<1	<1	<1	<1
1,2-DCE (total) (µg/l)	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	1	1	0.3 J
Ethylbenzene (µg/l)	0.02 J	0.02 J	0.02 J	0.02 J	0.05 J	0.04 J	0.07 J	<1	<1	<1	0.03 J	<1	<1	0.02 J	<1
PCE (µg/l)	<1	<1	0.1 J	<1	<1	<1	<1	0.05 J	3	<1	5	0.3 J	3	3	1
Toluene (µg/l)	<1	<1	0.1 J	0.03 J	0.2 J	0.1 J	0.2 J	<1	<1	<1	<1	<1	<1	<1	<1
trans-1,2-DCE (µg/l)	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	<1	0.2 J	<1	<1
TCE (µg/l)	<1	<1	<1	<1	<1	<1	0.3 J	<1	<1	<1	0.2 J	<1	0.4 J	0.4 J	0.7 J

Notes: DCA = Dichloroethane  
DCE = Dichloroethene  
TCA = Trichloroethane  
PCE = Tetrachloroethene  
TCE = Trichloroethene  
BDCM = Bromodichloromethane  
DUP = Duplicate sample

J indicates that the sample contains the compound at an estimated concentration.

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SOURCE: U. S. GEOLOGICAL SURVEY 7-1/2 MINUTE QUADRANGLE MAP FOR TELFORD, PA.

**LEGEND**

- ▲ NPWA WELL
- S-2
- RESIDENTIAL WELL
- R-X
- BOROUGH AND TOWNSHIP BOUNDARIES

**NOTES:**

1. DATA ARE IN µg/l
2. MULTIPLE RESULTS FOR ONE WELL AND COMPOUND INDICATE DUPLICATE.

0 600 1200 1800



SCALE: 1"=1200'

**FIGURE 4-7**

RESULTS OF ANALYSES OF GROUNDWATER FOR VOCs

North Penn Area 1  
Phase II RI/FS



AR300749

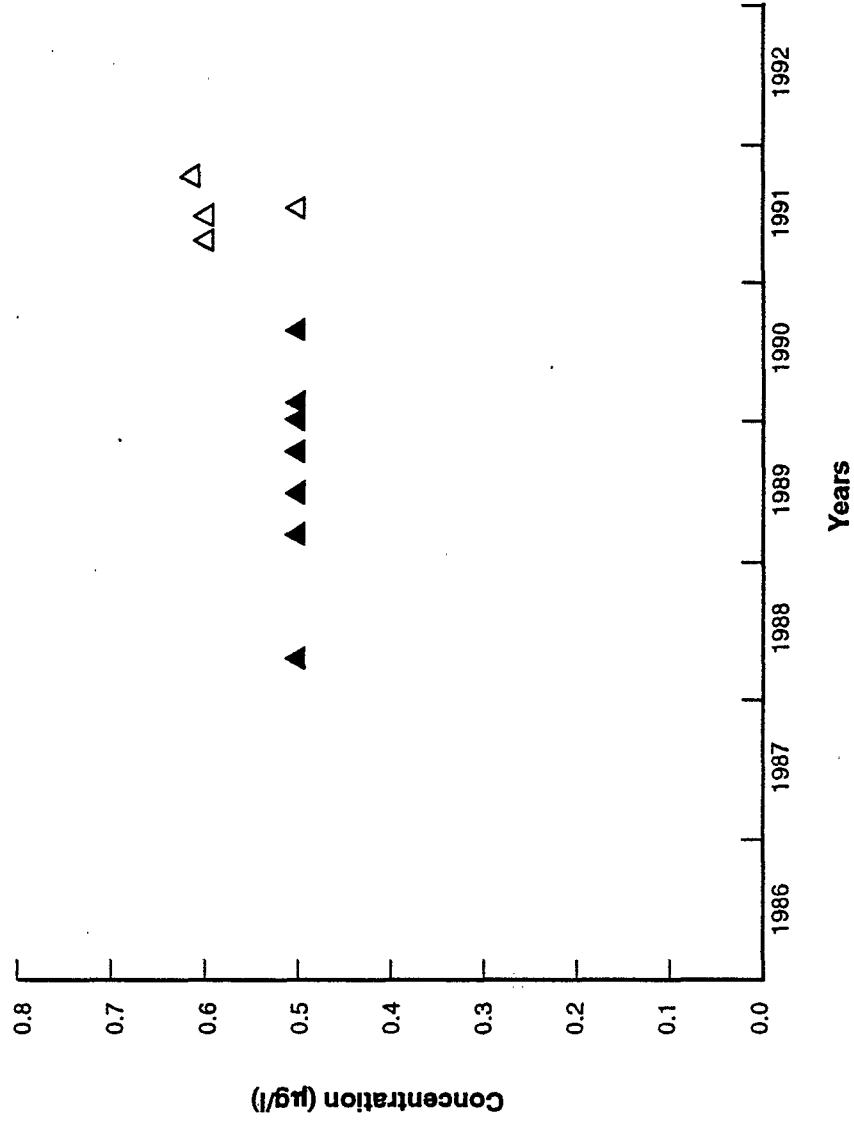
Long-term data on VOC contamination in groundwater are available from NPWA wells S-2, S-4, S-8, S-9, and S-10. These data are provided in Appendix A and are shown in Figures 4-8 through 4-12. PCE levels in well S-2 (Figure 4-8) have remained around 0.5 to 0.6  $\mu\text{g/l}$  for several years. PCE concentrations in well S-4 (Figure 4-9) have varied widely between about 2 and about 5  $\mu\text{g/l}$  and showed a general decline during 1991. These wells were not resampled during the RI.

Wells S-8, S-9, and S-10 were resampled during the RI; these recent data are not shown in the figures. PCE levels in well S-8 (Figure 4-10) remained at about the detection limit of 0.5  $\mu\text{g/l}$  for several years but in 1990 and 1991 rose to values on the order of about 1  $\mu\text{g/l}$ . The RI sampling detected a concentration of about 0.3  $\mu\text{g/l}$ , suggesting that the level may be declining. In well S-10, the PCE level has varied between about 0.5 and 1.5  $\mu\text{g/l}$  over time (Figure 4-11), higher than the 0.3  $\mu\text{g/l}$  detected during the RI. TCE and 1,1,1-TCA detected in well S-10 in earlier samplings were not detected during the RI.

In well S-9, the PCE level (Figure 4-12) was about 10 to 13  $\mu\text{g/l}$  during 1986 and 1987, the period of record for this well. The RI sampling detected only 5  $\mu\text{g/l}$ , suggesting that the concentration has declined since then. However, sampling of straddle-packed intervals in the well during the RI suggests that the concentration of 5  $\mu\text{g/l}$  may not be representative of the level of contamination in the well; this issue is discussed in more detail later when the straddle-packer test results are presented. Concentrations of TCE and 1,1,1-TCA in this well were on the order of 0.5  $\mu\text{g/l}$  in 1986 and 1987. TCE was detected at 0.2  $\mu\text{g/l}$ , and 1,1,1-TCA was not detected in the well when the entire well was sampled during the RI. The level of TCE detected in a sample from the entire well was about that detected during the straddle-packer tests.

Groundwater samples were collected from discrete intervals in the GKM well and in well S-9. The discrete samples were obtained by the use of straddle packers. The results of the packer sampling are summarized in Table 4-4. The packer sampling results indicate that the contamination extends throughout the total depths of both wells.

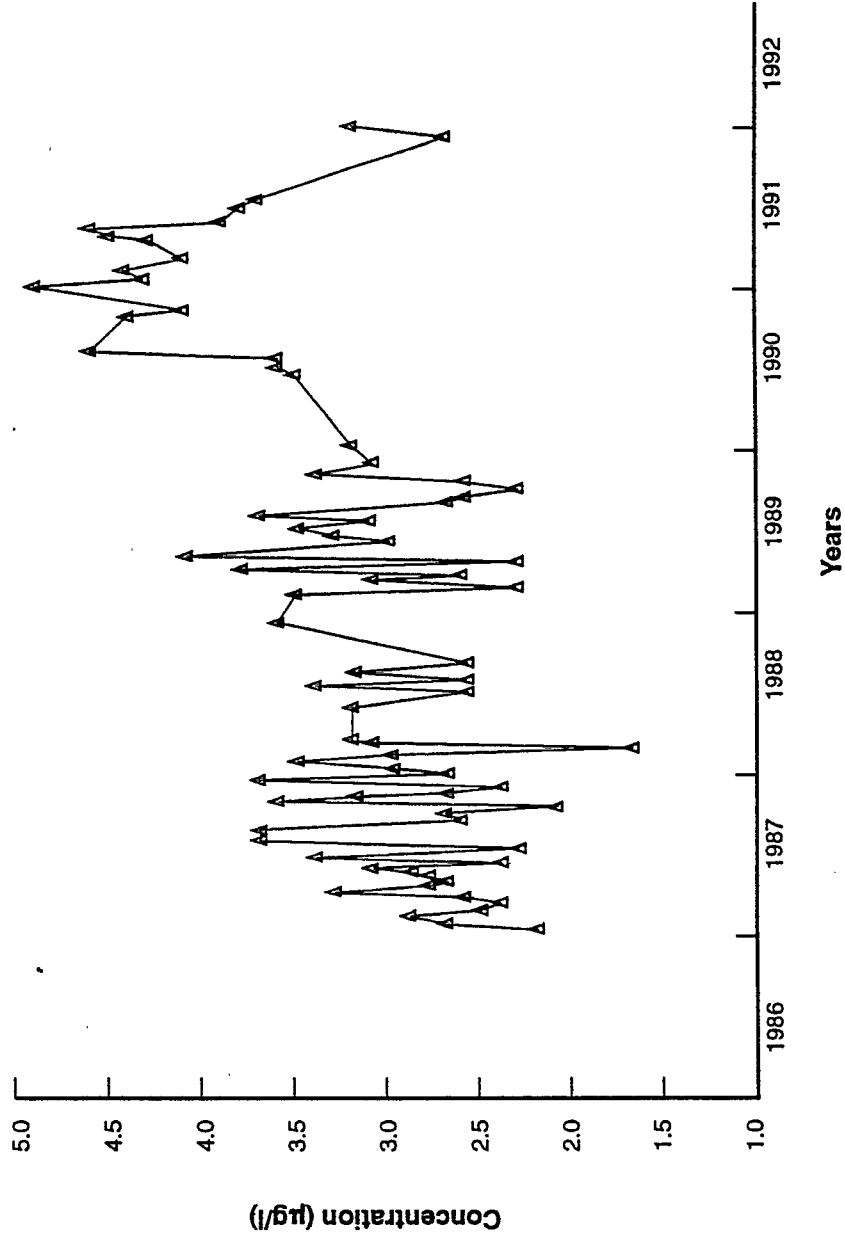
The samples collected from well S-9 exhibited contaminant levels ranging from 8 to 17  $\mu\text{g/l}$  of PCE. TCE also was detected in the samples from well S-9 wells at levels up to 0.3  $\mu\text{g/l}$ . The fact that the concentration of PCE was as high as 8 to 17  $\mu\text{g/l}$  in each of the five packed intervals suggests that the sample of the entire well may not have been representative of the level of contamination in the entire well. It is possible that the location of the pump used to purge the well drew water from the vicinity of a fracture or group of fractures that provided relatively clean water to the well. The well was sampled with a bailer lowered to this same interval. The original pump that was installed in the well and was probably used for the sampling in 1986 and 1987 was removed from the well so that packer testing could be performed. The pump used for the purging was set higher in the well, and the sample was obtained from a higher interval. This difference in sampling technique may account for the difference in the concentrations detected.



**Legend**  
△ PCE concentration in µg/l  
▲ PCE concentration below detection level of 0.5 µg/l

**Figure 4-8**  
**PCE CONCENTRATION OVER TIME**  
NPWA WELL S-2  
North Penn Area 1  
Phase II RI/FS

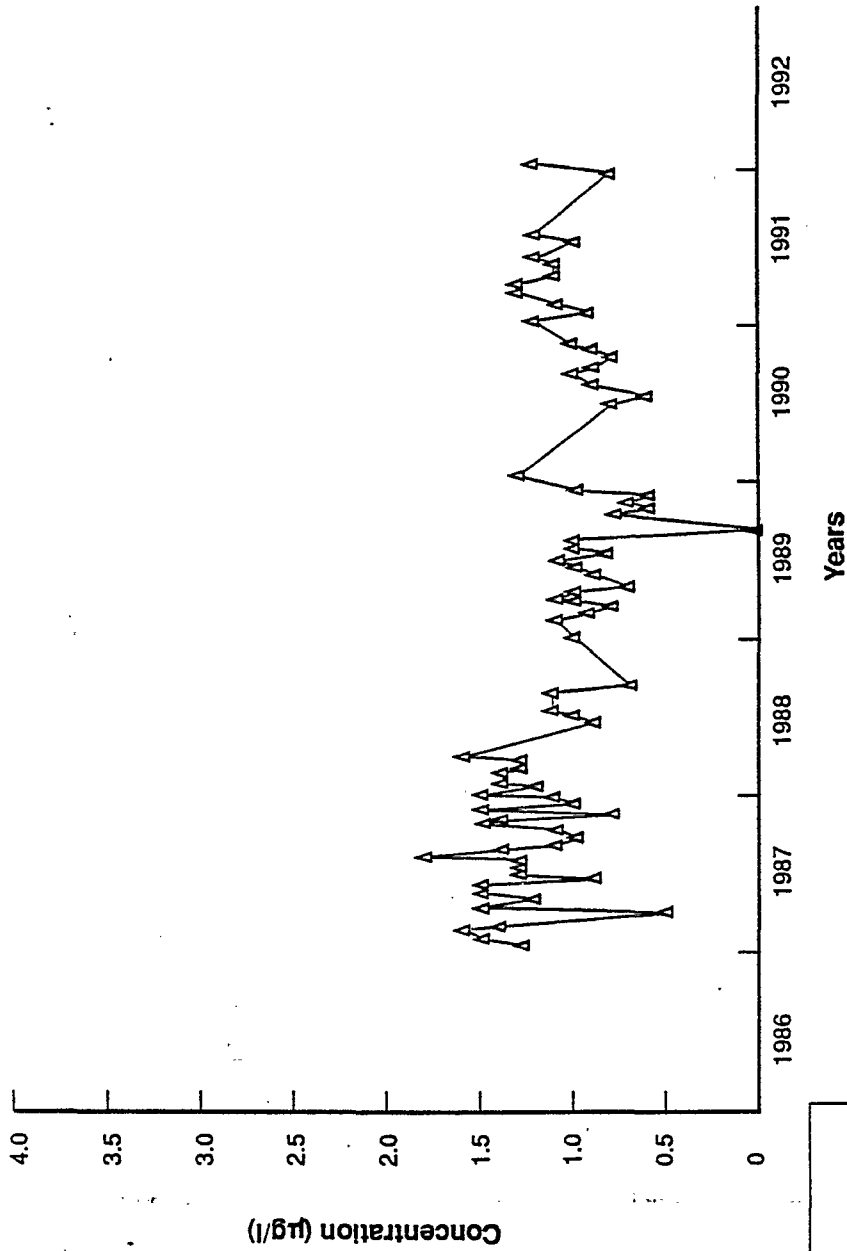




Legend  
△ PCE concentration in µg/l

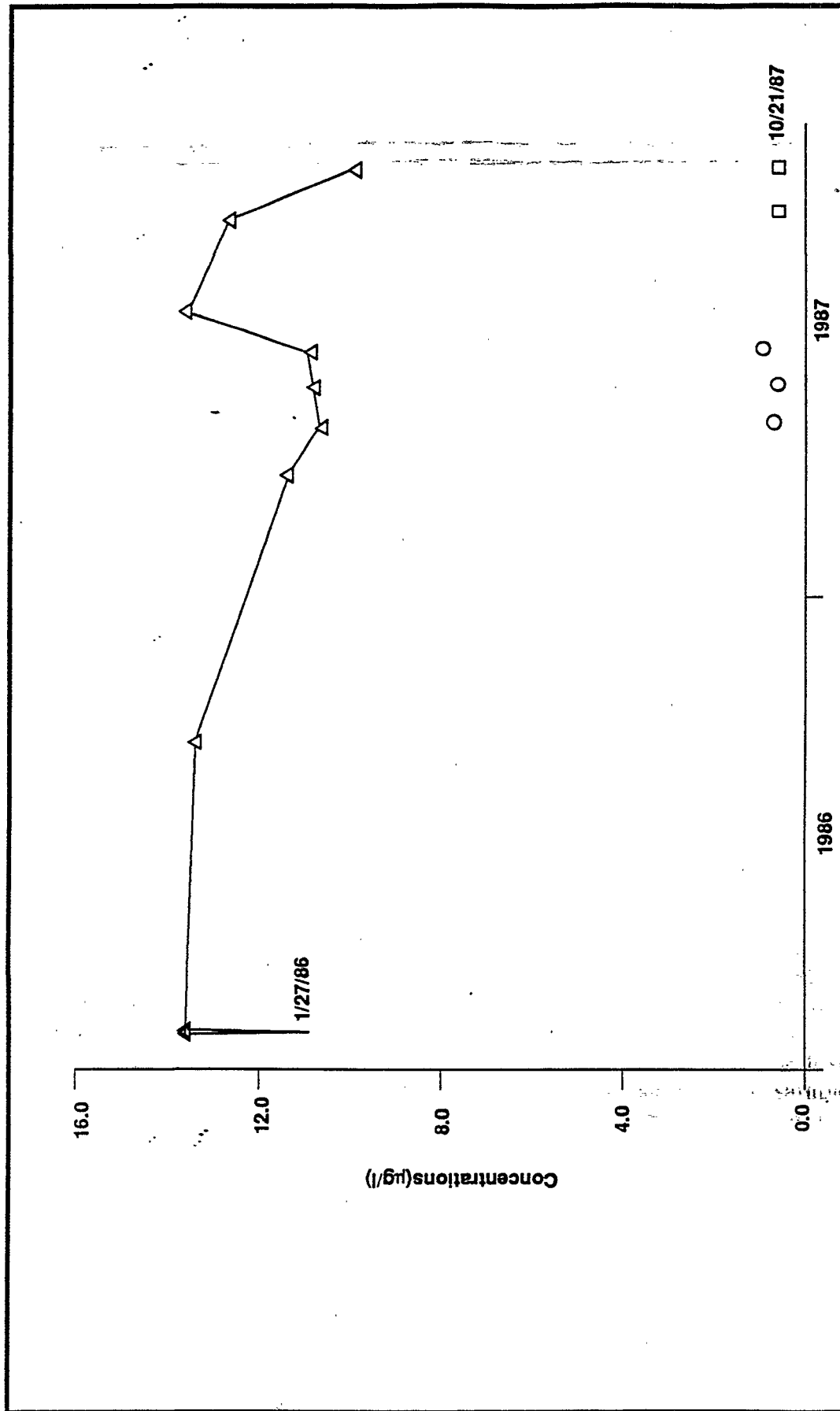
Figure 4-9  
PCE CONCENTRATION OVER TIME  
NPWA WELL S-4  
North Penn Area 1  
Phase II RI/FS





**Legend**  
△ PCE concentration in µg/l  
▲ PCE concentration below detection level of 0.5 µg/l

**Figure 4-11**  
**PCE CONCENTRATIONS OVER TIME**  
**NPWA WELL S-10**  
**North Penn Area 1**  
**Phase II RI/FS**



Legend

- 1,1,1-TCA Concentration in µg/l
- TCE Concentration in µg/l
- △ PCE Concentration in µg/l

Figure 4-12  
1,1,1-TCA, TCE, AND PCE  
CONCENTRATION  
OVER TIME-NPWA WELL S-9  
North Penn Area 1  
Phase II RI/FS



**Table 4-4  
ANALYSES OF ORGANIC COMPOUNDS IN GROUNDWATER SAMPLES OBTAINED  
DURING STRADDLE PACKER TESTS  
CLP RESULTS**

<b>Granite Knitting Mills Well</b>						
Sample No.	GKMS-3	GKMS-2	GKMS-4	GKMS-4 (DUP)	GKMS-5	GKMS-1
Depth (ft)	0-28	60-80	90-110	90-110	110-130	165-187
<b>Compound</b>						
1,1,1-TCA ( $\mu\text{g/l}$ )	1	0.3 J	0.4 J	0.3 J	1	1
1,1-DCA ( $\mu\text{g/l}$ )	0.3 J	0.1 J	<1	<1	0.3 J	0.4 J
1,2-DCE (total) ( $\mu\text{g/l}$ )	0.9 J	0.8 J	0.5 J	0.5 J	1	1
PCE ( $\mu\text{g/l}$ )	330	24	19	18	16	19
Toluene ( $\mu\text{g/l}$ )	1	1	0.3 J	0.4 J	0.3 J	5
TCE ( $\mu\text{g/l}$ )	6	0.9 J	0.7 J	0.6 J	0.9 J	1
TOC (mg/l)	6.9	9.6	4	8.3	23.6 J	34.8 J
<b>NPWA Well S-9</b>						
Sample No.	S9PTS-1	S9PTS-2	S9PTS-3	S9PTS-4	S9PTS-5	
Depth (ft)	77-97	130-150	158-178	195-215	245-265	
<b>Compound</b>						
PCE ( $\mu\text{g/l}$ )	9	8	10	10	17	
Toluene ( $\mu\text{g/l}$ )	0.08 J	0.1 J	<1	<1	<1	
TCE ( $\mu\text{g/l}$ )	0.2 J	<1	<1	<1	0.3 J	
TOC (mg/l)	7.6	3.5	2.3	4	8.1	
Notes: DCA = Dichloroethane                      TCE = Trichloroethene DCE = Dichloroethene                      TOC = Total organic carbon TCA = Trichloroethane                      DUP = Duplicate sample PCE = Tetrachloroethene						
J indicates that the sample contains the compound at an estimated concentration of less than the detection limit.						

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A groundwater sample obtained from the GKM well in 1986 reportedly contained 250  $\mu\text{g/l}$  of 1,1,1-TCA, 33.5  $\mu\text{g/l}$  of PCE, 12.2  $\mu\text{g/l}$  of TCE, and lower concentrations of other VOCs. The sample obtained from the packed intervals of the well during the RI contained almost no 1,1,1-TCA or TCE and variable levels of PCE (Table 4-4), the latter ranging from 330  $\mu\text{g/l}$  at the top interval to 16  $\mu\text{g/l}$  in a lower interval. These data suggest that the overall concentrations of VOCs in this well have declined over time.

The highest concentration of most VOCs in the GKM well were found in a sample collected from the uppermost interval (Table 4-4). During video logging of the well, water was observed to be running into the well borehole from fractures located below the bottom of the casing but above the water level in the well; the fractures were located at a depth of about 22 feet below the ground surface. The water observed to be entering the well from these fractures is probably surface infiltration that has passed through contaminated soil at Gentle Cleaners or other facilities, migrated to the bedrock surface, and entered fractures in the bedrock, finally traveling to the GKM well. Contamination entering the well from these fractures would contaminate the entire well. This contamination would probably be accomplished by diffusion and perhaps by density variations in the infiltrating water.

The sample of the entire GKM well (Table 4-3) was collected more than 24 hours into the 72-hour pumping test performed in the well. By that time the well had been purged of at least three well volumes. Samples also were collected periodically throughout the test and analyzed with the field GC to determine trends in contaminant levels. The results of these samples are presented in Table 4-5. The concentration-time data for PCE, TCE, and 1,1-DCE are shown in Figures 4-13 through 4-15. The data are plotted with equal time spacing between data points, although the time interval between samples actually increased as the test progressed. The gradual leveling off of the concentrations over time would still be apparent if the data were plotted with their true time spacing.

In all cases there is a sharp initial decline in concentration, followed by a gradual decline or a tendency to stabilize in most cases. An exception is an increase in 1,1-DCE late in the test. It should be noted that the PCE concentration had declined to a value on the order of 6  $\mu\text{g/l}$  by the end of the test and had dropped from an initial level of around 30  $\mu\text{g/l}$  to about 10  $\mu\text{g/l}$  within a few hours of the beginning of the test. This decline and general stabilization in PCE concentration suggests that the sample obtained from the entire well is representative of the quality of the groundwater in the vicinity of the well.

The concentrations of VOC contaminants detected in the groundwater samples collected in this RI are so low that there is no evidence that DNAPLs are currently affecting groundwater quality at the site. The concentrations observed are only a very

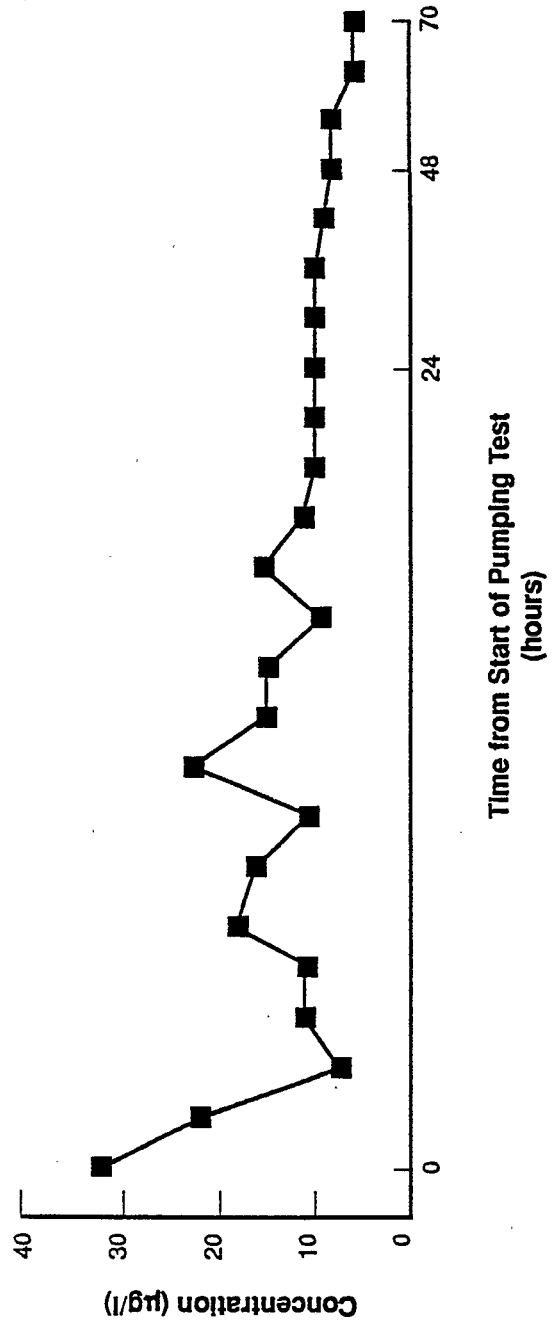


Figure 4-13  
PCE DETECTIONS  
GRANITE KNITTING MILLS PUMPING TEST  
North Penn Area 1  
Phase II RI/FS

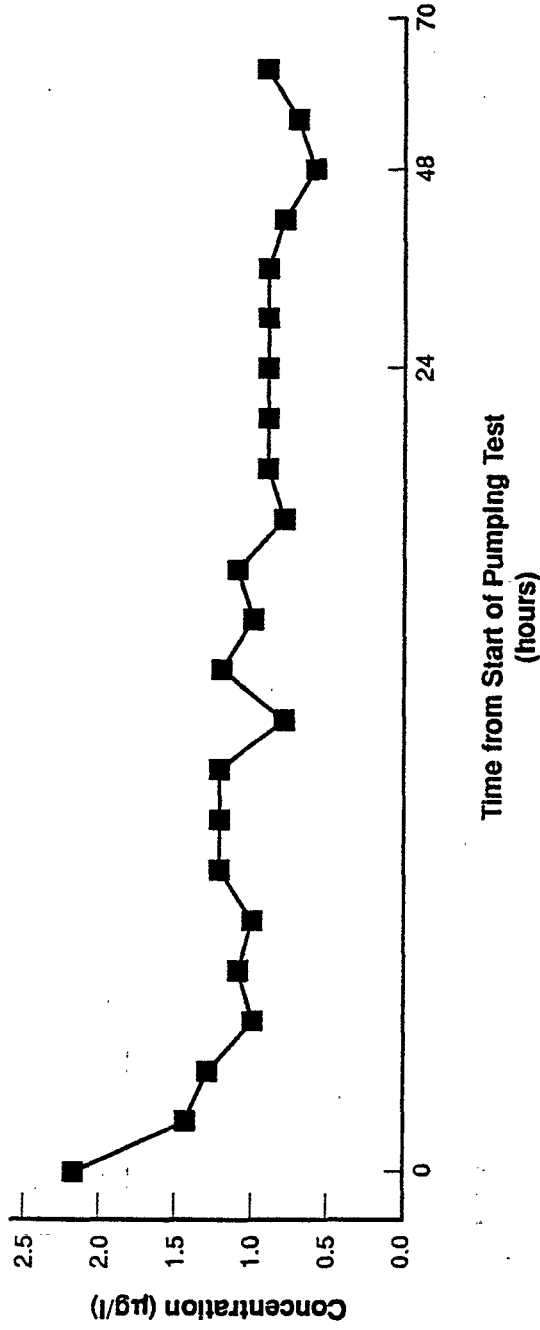


Figure 4-14  
TCE DETECTIONS  
GRANITE KNITTING MILLS PUMPING TEST  
North Penn Area 1  
Phase II RI/FS



T S19

T S20

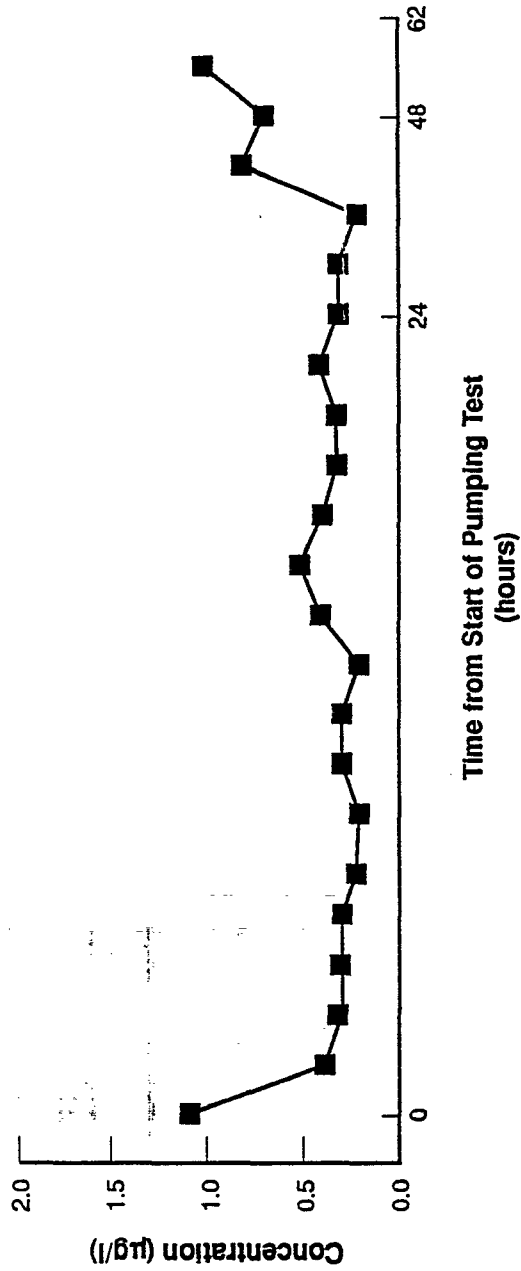


Figure 4-15  
DCE DETECTIONS  
GRANITE KNITTING MILLS PUMPING TEST  
North Penn Area 1  
Phase II FI/FS



**Table 4-5**  
**NORTH PENN AREA 1 GROUNDWATER RESULTS**  
**FIELD GC RESULTS**  
**GRANITE KNITTING MILLS PUMPING TEST (µg/l)**

Time Into Test (min.)	Sample Identification	Tetrachloroethylene	1,1-Dichloroethylene	cis-1,2-Dichloroethylene	Trichloroethylene
2	GKM PUT S1	32.0	1.1	10.6	2.2
5	GKM PUT S2	21.7	0.4	0.9	1.4
10	GKM PUT S3	7.3	0.3	0.5	1.3
15	GKM PUT S4	11.0	0.3	4.3	1.0
35	GKM PUT S6	18.0	0.2	4.7	1.0
60	GKM PUT S7	16.0	0.2	0.6	1.2
90	GKM PUT S8	10.3	0.3	0.6	1.2
150	GKM PUT S9	22	0.3	5.5	1.2
210	GKM PUT S10	14.8	0.2	0.5	0.8
390	GKM PUT S11	14.5	0.4	5.8	1.2
510	GKM PUT S12	9.2	0.5	5.5	1
630	GKM PUT S13	15	0.4	5.8	1.1
750	GKM PUT S14	11	0.3	4.9	0.8
990	GKM PUT S15	9.8	0.3	5	0.9
1,230	GKM PUT S16	10	0.4	4.7	0.9
1,470	GKM PUT S17	10	0.3	2.7	0.9
1,710	GKM PUT S18	10	0.3	2.7	0.9

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Table 4-5  
**NORTH PENN AREA 1 GROUNDWATER RESULTS**  
**FIELD GC RESULTS**  
**GRANITE KNITTING MILLS PUMPING TEST ( $\mu\text{g/l}$ )**

Time Into Test (min.)	Sample Identification	Tetrachloroethylene	1,1-Dichloroethylene	cis-1,2-Dichloroethylene	Trichloroethylene
1,950	GKM PUT S19	9.9	0.2	2.6	0.9
2,550	GKM PUT S20	8.7	0.8	4.3	0.8
2,910	GKM PUT S21	7.4	0.7	3.9	0.6
3,270	GKM PUT S22	7.4	1	3.6	0.7
3,690	GKM PUT S23	5.6	ND	3.8	0.9
4,110	GKM PUT S24	5.4	ND	1.9	ND
ND--Not Detected					

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AR300762

small percentage of the solubility limits of the VOCs of concern. During the periodic sampling of the groundwater being discharged from the GKM well, the concentration of PCE stabilized but at a concentration of only about 6  $\mu\text{g/l}$ , well below the solubility limit of TCE. Therefore, the presence of DNAPLs was not indicated by this sampling.

In summary, contamination by VOCs of the groundwater at the site still exists but may have declined in concentration over time. PCE was detected at or above the MCL of 5  $\mu\text{g/l}$  only in well S-9. PCE was detected, in this investigation, at a lower concentration in production well S-10 and not detected at all in production well S-8. PCE was detected in NPWA sampling in 1991 at less than 1  $\mu\text{g/l}$ . Contamination from the Area 1 plume may have affected a residential well southwest of the site. High concentrations of VOC contaminants are still entering the groundwater, at least at the location of the GKM well, although they are diluted in the aquifer to much lower levels.

There is no definitive evidence for the presence of DNAPLs in the groundwater at the site. Even a concentration of PCE at 330  $\mu\text{g/l}$ , detected in packer sampling at the GKM well, is only 0.2 percent of the solubility of PCE. However, it is known that a spill of PCE occurred at Gentle Cleaners that probably introduced DNAPLs to the subsurface, and the presence of DNAPLs would explain why PCE was detected throughout the entire depths of the GKM well and well S-9, although at concentrations that are only a small fraction of the solubility of PCE.

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**Section 5**  
**Contaminant Fate and Transport**

AR300764

## Section 5 Contaminant Fate and Transport

### 5.1 Introduction

This section describes the potential transformations and transport of the contaminants that have been detected at the site. For each medium, the processes that control a chemical's fate and transport are described in general. The contaminants of concern at the site are listed, along with relevant chemical and physical properties and property definitions. On the basis of the results of the site investigation, the anticipated behavior of each contaminant in the media of concern is discussed.

### 5.2 Media of Concern

The media of concern for the site include surface soil, subsurface soil, groundwater, and surface water. Contaminants have been detected in both soils and groundwater.

#### 5.2.1 Surface Soil

Surface soil is commonly defined as the top zero to 2 feet of soil. Contaminants enter the surface soil when they are spilled during transport or storage. Their persistence in the surface soil is a function of the chemical and physical properties of the soil and the contaminants. Soil is made up of three phases: the solid soil particles, soil water, and soil gas; the latter is predominantly air in surface soil. Soil properties that affect the distribution of a contaminant within these three phases include porosity, percentage of organic carbon, moisture content, Ph, particle size, and mineralogy.

In general, organic contaminants are held in the soil matrix by adsorption to the soil particles. Except for soils with significant clay content, the contaminant sorbs to the organic carbon portion of the soil; the degree of sorption increases with increasing organic carbon content.

Transformation processes that may occur are associated with the portion of the contaminant that is present in the soil water. If biological degradation takes place, it will be aerobic, since oxygen will readily diffuse down into the soil. Biodegradation requires the presence of nutrients, water, acclimated microbes, and sources of carbon and energy for the organisms. Soil environments that are most conducive to microbial growth are moist soils with high organic carbon content and warm temperatures. Even if conditions are ideal for the growth of organisms, biodegradation may not occur; degradation potential is contaminant-specific. Abiotic degradations that may decrease contaminant concentrations in soil include photolysis at the soil surface and hydrolysis in the soil water.

The transport processes that apply to surface soil contaminants are volatilization and leaching. Volatilization occurs when contaminants partition to the gas phase from the water phase. The vapors diffuse upward to the soil surface, where they are removed by wind. Volatilization will increase with evaporation of soil water from the soil surface. Contaminants also may diffuse downward to the water table where they dissolve into the groundwater. Leaching occurs when precipitation infiltrates through the surface soils, carrying the dissolved contaminant to the subsurface soils.

### **5.2.2 Subsurface Soil**

Transformations that are relevant to subsurface soils are biological and abiotic degradation. Potential biological degradation will more likely be anaerobic as the depth below surface increases and oxygen is consumed in any aerobic reactions. Soil moisture generally increases with depth, increasing the amount of contaminant that is present as dissolved in water.

Contaminated surface soil potentially serves as a source of contamination of subsurface soil and groundwater. Transport into, through, and out of the subsurface soil occurs via several processes:

- Gas-phase advection driven by the difference in density between the contaminant in the gas phase and ambient soil gas
- Gas-phase diffusion due to concentration differences of contaminant in soil gas
- Liquid-phase advection of contaminant dissolved in infiltrating water by leaching
- Liquid-phase diffusion
- Liquid-phase migration of non-aqueous phase liquids

In general, the most significant process is leaching. As water infiltrates the contaminated soil, the contaminant is transferred to the infiltrating water. When the water contacts previously uncontaminated soil, the contaminant redistributes to soil particles. The rates of migration of different contaminants by this process vary greatly depending on the rate of exchange between soil and water, infiltration rates, and soil and contaminant properties. Non-aqueous phase liquids will move under the influence of gravity downward through unsaturated soils.

At the site, the water table is below the transition from soil to bedrock. If contaminated water reaches the bottom of the soil, it will enter the fractures in the rock and move down into the groundwater.

### 5.2.3 Groundwater

Once contaminants reach the groundwater, they will be reduced in concentration as they are diluted by upgradient water. As long as the contaminant concentration is lower than its solubility limit in water, it will be transported through the aquifer as a dissolved phase with the groundwater, but it will lag behind the bulk groundwater velocity, because of adsorption to the aquifer matrix. The greater the degree of adsorption, the slower the contaminants migrate, because their progress is retarded by the exchange process. Adsorption that is a function of the amount of organic material present is probably reduced in magnitude because less organic material is available at greater depths in the subsurface and in the groundwater system.

Other processes that potentially influence concentrations as the contaminant migrates through the aquifer are dispersion, volatilization, and biological and abiotic degradation. Dispersion acts to spread the contaminant out as a result of diffusion and hydraulic mixing. The greater the degree of dispersion, the lower the concentration is along the centerline of a migrating plume and the more broadly the contaminants are distributed. Volatilization would be a significant loss mechanism for volatile organic compounds in groundwater; however, this is generally not the case. Volatilization is limited by the diffusion or dispersion of volatile contaminants through the groundwater to the water-air interface, followed by diffusion to the soil surface. At the low velocities typical of groundwater, volatilization is not significant.

The potential for biological degradation is extremely site-dependent. The controlling factors are the same as those listed for degradation in soils. Although hydrolysis rates are slow, they may be significant in the time scales associated with groundwater movement.

Non-aqueous phase liquids will not mix immediately with groundwater. Rather, they will remain a separate phase and either float on the water table if lighter than water or sink downward if denser than water and migration pathways are available. If denser than water, the non-aqueous phase liquids will move under the influence of gravity rather than under the influence of groundwater flow.

## 5.3 Chemical and Physical Properties

Table 5-1 contains a list of the contaminants that were present at detectable quantities in the soil and groundwater samples at the site. The chemical and physical properties that affect fate and transport in the environment are listed for each contaminant. Definitions of each property, as well as information that the properties provide about fate and transport, follow.



Table 5-1  
 PHYSICAL AND CHEMICAL FATE DATA

Chemical Name	Synonyms	Molecular Weight (gm/mole)	Water Solubility (mg/l) @ 25°C	Vapor Pressure (mm Hg) @ 25°C	Henry's Law Constant (atm-m <sup>3</sup> /mole)	K <sub>oc</sub>	Hydrolysis Half-Life (years) @ 25°C
Tetrachloroethene	PCE Perchloroethylene Tetrachloroethylene	165.82	150	18.5	1.49E-2	210	Not significant (N/S)
Trichloroethene	TCE Trichloroethylene	131.4	1100	69.0	1.03E-2	104	N/S
1,1,1-Trichloroethane	TCA	133.42	1495	123.7	8.0E-3	179	1.1
cis-1,2-Dichloroethene	cis-1,2-DCE cis-Dichloroethylene	96.94	3500	200 @ 35°C	3.37E-3	49	N/S
trans-1,2-Dichloroethene	trans-1,2-DCE trans-1,2-Dichloroethylene	96.94	6300	340	6.72E-3	58.9	N/S
1,1-Dichloroethene	1,1-DCE 1,1-Dichloroethylene Vinylidene Chloride	96.94	2500	591	3.01E-2	150	N/S
Acetone	Methyl ketone	58.08	Infinite	266	0.000040	0.37	Not available (N/A)
Benzene	Benzol	78.11	1,800	95.2	0.00548	49	Not significant (N/S)
Bromodichloromethane	Dichlorobromomethane	163.83	4,500	50 @ 20°C	0.0024	62	1,374
Carbon disulfide	Carbon bisulfide	76.13	1,700	297.5 @ 20°C	0.0133	240	N/A

Table 5-1  
PHYSICAL AND CHEMICAL FATE DATA

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Chemical Name	Synonyms	Molecular Weight (gm/mole)	Water Solubility (mg/l) @ 25°C	Vapor Pressure (mm Hg) @ 25°C	Henry's Law Constant (atm-m <sup>3</sup> /mole)	K <sub>oc</sub>	Hydrolysis Half-Life (years) @ 25°C
Chloroform	Methyl trichloride	119.38	9,300	160 @ 20°C	0.0029	44	N/A
Chloromethane	Methyl chloride	50.48	7,400	3,789 @ 20°C	0.00882	25	1.2
Ethylbenzene	Ethylbenzol	106.17	206	12 @ 30°C	0.0066	95	N/S
Methylene chloride	Dichloromethane	84.93	16,700	348.9 @ 20°C	0.0020	8.7	688
Toluene	Methylbenzene	92.14	524	22 @ 20°C	0.0067	115	N/S

Sources: Montgomery and Welkom (1990) except hydrolysis rates from SRI (1982)

Howard, Philip H., *Handbook of Environmental Fate and Exposure Data For Organic Chemicals*, Lewis Publishers, Inc., 1990.

Risk\*Assistant (Version 1.0C), Chemical Properties Database, EPA, Thistle Publishing, 1992.

Hazardous Substances Data Bank (HSDB), online database, National Library of Medicine, Bethesda, MD, Micromedex, Inc., Denver, CO, 1992.

Jeffers, Peter M., "Homogeneous Hydrolysis Rate Constants for Selected Chlorinated Methanes, Ethanes, Ethenes, and Propanes," *Environ. Sci. Technol.*, Vol. 23, 965-969, 1989.

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Principal properties that influence contaminant migration include the solubility, vapor pressure, Henry's Law Constant (which describes volatilization), and carbon/water partition coefficient (which describes adsorption). The water solubility of a chemical is the maximum concentration of a contaminant that can dissolve in pure water at a given temperature. Highly soluble chemicals can be rapidly leached from contaminated soil and are relatively mobile in groundwater. In general, volatile organic compounds (VOCs) have rather high water solubilities relative to their water quality criteria; solubilities for all but one of the site contaminants are greater than 1,000 mg/l. The exception is PCE, which has a solubility of 150 mg/l.

The degree of volatilization of a compound may be indicated by its vapor pressure and water solubility. Highly water-soluble chemicals generally have lower volatilization rates from water, unless they also have higher vapor pressures. The Henry's Law Constant combines vapor pressure with solubility and is more appropriate than vapor pressure alone for estimating releases of organic compounds from water to air. It is defined as the concentration of contaminant in the vapor phase, divided by that in the aqueous phase. Compounds with Henry's Law Constants greater than  $10^{-3}$  atmosphere-cubic meters per mole ( $\text{atm}\cdot\text{m}^3/\text{mol}$ ) (e.g., many VOCs) can be expected to readily volatilize from water to air; those with values ranging from  $10^{-3}$  to  $10^{-5}$   $\text{atm}\cdot\text{m}^3/\text{mol}$  are associated with possibly significant volatilization. As their name indicates, VOCs are highly volatile.

The carbon/water partition coefficient ( $K_{oc}$ ) is often used to estimate the extent to which a chemical will partition between the organic material in the soil and the water. Chemicals with low  $K_{oc}$  values (less than 10 milliliters per gram) are found mainly in the water phase, while chemicals with high  $K_{oc}$  values tend to adsorb. The distribution coefficient ( $K_d$ ) is defined as the product of the  $K_{oc}$  and the fraction of organic carbon.

Persistence is a measure of how long a chemical will exist in air, water, or soil. Phase transfer and chemical and biological transformation are major forms of removal processes. Persistence is expressed as a half-life ( $t_{1/2}$ ) in days. A short half-life indicates a lower level of concern. In general, metals and pesticides are persistent in the water and soil media. Degradation can occur by both chemical and biological means. Chemical degradation can occur through such processes as hydrolysis and photolysis. Biological degradation occurs among organic compounds, with volatile organic compounds typically being more biodegradable than semivolatile organic compounds. Rates of degradation are available in the literature but vary greatly according to the nature and conditions of the experiments. Hydrolysis rates for the chemicals of concern are provided in Table 5-1.

## 5.4 Fate and Transport

The results of the field investigation and nature and extent of contamination are combined with fate and transport concepts and contaminant properties to produce a conceptual model of contaminant transport at the site. The data are discussed by medium and area.

### 5.4.1 Soil

#### *Gentle Cleaners*

Analyses of surface soil samples detected only one case of contamination: 23  $\mu\text{g}/\text{kg}$  of PCE. No other chemicals were detected at the surface. PCE has a high Henry's Law Constant; it will volatilize easily from surface soils. It is likely that the majority of the PCE, which was the source of deeper contamination, has volatilized or leached downward to the subsurface soils.

From the 8 borings, 12 subsurface samples detected levels of PCE ranging from nondetected to 1,449  $\mu\text{g}/\text{kg}$ . The exception was one sample from a boring, which gave a PCE level of 300,000  $\mu\text{g}/\text{kg}$ . The 300,000  $\mu\text{g}/\text{kg}$  sample was taken at a depth of 8 to 10 feet and was located at the place where solvents may have been spilled or dumped.

Approximately 6 feet to the southeast of the 300,000- $\mu\text{g}/\text{kg}$  sample, the property drops in elevation by 4 feet at a wall. This adjacent yard showed probable stains in aerial photographs. Soil borings on this lower property, at 5 feet from the wall, gave PCE levels of 100 and 1,449  $\mu\text{g}/\text{kg}$  (Contract Laboratory Program [CLP] and field results, respectively). The surface sample giving a PCE concentration of 23  $\mu\text{g}/\text{kg}$  was taken 10 feet from the wall. It is likely that surface water or groundwater has transported the PCE from the area of the highest levels downgradient to the lower-elevation property.

The other compound that was identified in soil borings at this location was a single TCE detection at 1  $\mu\text{g}/\text{kg}$ , located at the same boring and depth as the 300,000  $\mu\text{g}/\text{kg}$  PCE level. One  $\mu\text{g}/\text{kg}$  is an extremely low level of TCE. It is possible that this is a biological degradation product of the PCE. With the 300,000  $\mu\text{g}/\text{kg}$  PCE, oxygen would quickly be depleted from the subsurface soil, and conditions would be anaerobic near the contamination. TCE is the anaerobic degradation product of PCE, and the reaction is thermodynamically very favorable. The possibility exists for further TCE degradation to 1,1-DCE or 1,2-DCE (both cis- and trans-) if the TCE levels increase. This transformation is not as favorable as the PCE-to-TCE reaction.

The soils at the Gentle Cleaners site are silts with little clay and a trace of gravel. The depth to bedrock is approximately 10 feet behind the cleaners and 5 feet at the lower property. The combination of low clay content and relatively shallow soil indicates that the possibility exists for contamination to leach down to the groundwater. PCE is mobile in soils and will tend to move downward with infiltration. The high PCE level

was detected at the soil just above the bedrock, so the contaminant may have already migrated down from the surface. The effect of PCE leaching into groundwater is investigated later in this section using a leaching model.

A concentration of PCE on the order of 300,000  $\mu\text{g}/\text{kg}$  in soil suggests that a non-aqueous phase liquid has contacted the soil. This concentration is well above the solubility limit of PCE in water. This is consistent with the reported spill of 75 gallons of PCE at this facility. The non-aqueous phase liquid would have migrated downward through the soil to the bedrock, ultimately reaching the groundwater.

### ***Granite Knitting Mills***

The primary contaminant detected in soils at the Granite Knitting Mills area was PCE. Levels of PCE in surface soils were 32 and 1,000  $\mu\text{g}/\text{kg}$  from the CLP and field gas chromatograph (GC) analyses, respectively. No other contaminants were detected in the surface soils.

PCE in the subsurface was detected at 6,900  $\mu\text{g}/\text{kg}$ , at a depth of 6 to 8 feet. The next highest level was 152  $\mu\text{g}/\text{kg}$ ; the remainder of the levels were below 100  $\mu\text{g}/\text{kg}$ . As described above, PCE is mobile in soils and is probably leaching downward with infiltration. The rate at which PCE will potentially migrate down will be slowed at the GKM site because the soil is covered with asphalt.

Secondary contaminants detected in the soil samples were trans-1,2-DCE, cis-1,2-DCE, and TCE. These were detected in the same sample as the highest PCE level. All of these compounds are VOCs, expected to volatilize from and leach readily through soil. The fact that the contaminants were detected in the same sample can be explained by at least two processes. The PCE may be degrading biologically to TCE and DCE, or the original source of the PCE also may have been the source of the other contaminants. Since the mill was known to have used several types of solvent and dyes, the latter explanation is possible.

### ***Parkside Apartments***

The primary contaminant in this area was PCE, with levels in the subsurface to 1,787  $\mu\text{g}/\text{kg}$  (GC and 120  $\mu\text{g}/\text{kg}$  CLP data) from one sample. The surface PCE level was only 3  $\mu\text{g}/\text{kg}$  (field GC). The PCE concentrations appear to be the highest close to the building, decreasing away from the building; however this is not as evident in the direction toward the intermittent stream. Other contaminants detected on the site were TCE and cis- and trans-1,2-DCE. These were present only in samples adjacent to the building.

The decrease in concentrations away from potential sources and the lower levels at the surface are typical behavior for these compounds. It is likely that concentrations are decreasing as the contaminants move through the soil away from their original sources.

Causes of this reduction are dilution, dispersion, and possibly biodegradation. In the Parkside Apartments area, this direction is toward the intermittent stream to the northwest. As water infiltrates the facility property, it may initially move into the subsurface, to resurface at the streambed when the stream is flowing.

### *Lexco Engineering*

No VOCs were detected in the soil samples at this location.

### *Standard Terry Mills*

No VOCs were detected by the field GC in any of the soil samples from this facility. PCE at an estimated concentration of 7  $\mu\text{g}/\text{kg}$  was detected in one CLP sample.

## **5.4.2 Groundwater**

The results of the analyses of groundwater samples were described in Section 4.3. The CLP data showed detectable quantities of PCE in three of the wells tested; S-9, the Granite Knitting Mills (GKM) well, and a Souderton Borough well. The levels in these wells were 1 to 5  $\mu\text{g}/\text{l}$ . The wells also contained TCE (0.2  $\mu\text{g}/\text{l}$ ) and cis-1,2-DCE at 1  $\mu\text{g}/\text{l}$ . One well showed 0.9  $\mu\text{g}/\text{l}$  of 1,1,1-TCA.

These levels indicate that at least some soil contamination has leached down into the aquifer. All of the compounds detected are soluble in water and are expected to be transported fully dissolved in the groundwater. The concentrations will be diluted by the processes discussed in Section 5.2. However, if they enter the aquifer as non-aqueous liquids they can travel downward rapidly because of their high density and low viscosity. Soils data from Gentle Cleaners suggest the likelihood that such liquids were present in the soil at some time, although no evidence was found in the groundwater.

Contaminants percolate or are leached through the soil layer (which is thin at the site), enter the fractures in the rock, and move down to the water table. At the water table, the contaminants enter the groundwater system. If the water table is a subdued representation of the surface topography and no pumping influences the groundwater flow, groundwater and dissolved contamination move in a generally south-to-southeast direction from Gentle Cleaners and Granite Knitting Mills and in a northwest direction from Parkside Apartments toward the unnamed tributary to Skippack Creek. The amount of fracturing of the bedrock and the extent to which these fractures are interconnected determines the extent to which the water table assumes a subdued representation of the surface topography. The tendency for groundwater to move in these directions would be influenced by pumping at NPWA wells S-8 and S-10, the generally northeast-to-southwest trend of fractures in the bedrock in the area, and the course of the unnamed tributary. Pumping of local domestic wells may influence the flow direction to some degree, but this is not believed to occur. Contaminated

groundwater either is removed from the hydrogeologic system by pumping wells or by discharge into the unnamed tributary.

It is possible that some biological degradation is taking place in the groundwater. Although organic carbon levels in bedrock are extremely low, microbes are known to be indigenous to groundwater environments. Measurements of the dissolved organic carbon in the groundwater, as well as dissolved oxygen concentration, would aid in determining whether biodegradation is occurring. If conditions are anaerobic, then the degradation products of PCE are TCE and DCE. The more chlorinated the compound, the more oxidized the compound is, and anaerobic dechlorinations are more favored. As the degree of chlorination decreases, so does the driving force for anaerobic dechlorination. Therefore, PCE is the most likely of the chlorinated ethenes to degrade in anaerobic conditions. Under aerobic conditions, PCE will not be biologically degraded.

Packer tests were performed on the GKM and S-9 wells. At the GKM well, PCE concentrations were 16 to 24  $\mu\text{g/l}$  from discrete intervals below the water table, but water entering the well from fractures above the water level contained PCE an order of magnitude greater, at 330  $\mu\text{g/l}$ . Other contaminants detected were TCE, 1,1-DCA, TCE, and cis-1,2-DCE; levels of these VOCs were only 0.1 to 6  $\mu\text{g/l}$ . As expected, the VOC concentrations are greatly diluted after entering the groundwater. PCE levels in S-9 ranged from 8 to 17  $\mu\text{g/l}$ , with TCE levels ranging from not detected to 0.3  $\mu\text{g/l}$ .

The analyses of groundwater indicate that contamination has reached the aquifer below the site. Pumping from the GKM well appeared to dilute the concentrations in the well, indicating less-contaminated groundwater surrounding the well. Contaminated soil was detected at both the GC and the GKM sites; the source of the contamination in the aquifer has not been definitively determined from this investigation. The contamination at Gentle Cleaners is upgradient of the GKM well, and it is possible that contamination from this source has traveled downgradient to the well via surface runoff and groundwater movement.

### **5.4.3 Surface Water and Sediment**

The boring at the Granite Knitting Mills area, which gave a PCE concentration of 1,000  $\mu\text{g/kg}$  at the surface, was located directly upgradient of a storm sewer grate. To check for contaminant dissolving in surface water or soil and entering the sewer, the outfall to the intermittent creek, which feeds the Skippack Creek, was sampled when it was flowing. No traces of VOCs were detected. No sediment samples were analyzed.

## 5.5 Transport Modeling

### 5.5.1 Introduction

As described in the preceding discussion on soil contamination at the site, there is a potential for contaminants in the soil to leach from the soil and migrate into the groundwater. In this case, residual chemicals of concern in the soil represent a potential source of contamination to groundwater. Modeling was performed to estimate the effects of this residual contamination on the quality of groundwater at the site.

### 5.5.2 Approach

The model commonly known as the "Summers" model (Summers et al., 1980) was used for the simulation. Because of its simplicity and ease of application, the Summers model is often used as a screening tool to determine quickly if a potential exists for groundwater contamination at a site and if a more complex model may be needed. It is also used when few data are available to adequately define the site lithologic, hydrogeologic, and transport parameters. The Summers model was selected for this application for both of these reasons. In particular, it is extremely difficult to define the transport properties of a fractured bedrock aquifer adequately to make a more-complex model useful and representative. Hence, the screening approach was selected.

The Summers model is a simple mass-balance model of the following form:

$$C_{gw} = \frac{Q_p * C_p + Q_a * C_a}{Q_a + Q_p}$$

- Where:
- $C_{gw}$  = concentration of the contaminant in the aquifer (mg/l)
  - $Q_p$  = volumetric flow rate transporting contamination through the vadose zone into the aquifer = area of the contaminated soil \* recharge rate (ft<sup>3</sup>/yr)
  - $C_p$  = concentration of the contaminant in the water infiltrating through the vadose zone (soil water) to the aquifer (mg/l)
  - $Q_a$  = volumetric flow rate in the aquifer (ft<sup>3</sup>/yr)
  - $C_a$  = upgradient concentration of the contaminant in the aquifer (mg/l)



The concentration of the contaminant in the soil water is determined by using the relationship:

$$K_d = C_s / C_p$$

Where:  $K_d$  = distribution coefficient describing the partitioning of the contaminant between soil and water (ml/gm)

$C_s$  = concentration of the contaminant in the soil (mg/kg)

The Summers model has several restrictive assumptions. The only fate and transport process accounted for is dilution in the aquifer immediately underlying the source. Although the combination of equations described above also accounts for adsorption (i.e., the  $K_d$  term) it determines only the concentration of the contaminant distributed from the source (i.e., the contaminated soil) to the soil water initially and does not account for retardation of the contaminant as it migrates to the water table.

There are several other assumptions. The source is assumed to be constant over time, so the decline in the concentration in the soil at the source is not accounted for. The simulated migration to the aquifer assumes no hydrodynamic dispersion or volatilization; therefore, the concentration of a contaminant reaching the water table has not been reduced by either of these processes. The contamination in the infiltrating soil water is assumed to be distributed uniformly over the area of the source. Finally, the contaminant reaching the water table is assumed to be uniformly mixed throughout the full thickness of the aquifer.

The values for hydraulic conductivity, recharge, aquifer thickness, and  $K_d$  are estimated on the basis of site knowledge to the extent possible and on literature values.

The use of the Summers model represents only a screening approach to modeling the site. Use of data that are not site-specific, along with a simplistic model, introduces some uncertainty into the calculated results. This uncertainty could be reduced by using a more complex model and site-specific measurements.

### **5.5.3 Conceptual Model**

The effect of the residual contamination by PCE in the soil at Gentle Cleaners was simulated. The assumption was made that contamination at this facility was leached from the soil by soil water derived from infiltrating precipitation. The contaminant concentration in the soil water was assumed to be in equilibrium with the concentration in the soil. The contaminated soil water was then mixed with groundwater in the underlying aquifer to obtain a concentration in the groundwater.

Table 5-2 contains values for the input parameters for the model and the results of the calculations. The area containing the contaminated soil was assumed to be 412 ft<sup>2</sup>, the area of the backyard at Gentle Cleaners. The recharge rate was assumed to be one-third of the average precipitation rate of 45 inches per year, or 1.25 feet per year; this value for recharge was not measured in the field but is typical. The product of these two parameters yielded a value of 515 ft<sup>3</sup> per year of recharge over the backyard of the facility.

The infiltrating contaminated soil water was assumed to mix uniformly with the groundwater in the bedrock. However, the thickness of the aquifer with which the contaminated soil water mixes is uncertain. A method to estimate this thickness described by Salhotra et al. (1990) was used:

$$H = (2a_v L)^{1/2} + B(1 - \exp(-LQ/Vn_e B))$$

Where:	H =	Mixing zone thickness	
	L =	Length of the facility parallel to groundwater flow	= 25 ft
	a <sub>v</sub> =	Vertical dispersivity = 0.006 x L (Salhotra et al., 1990)	= 0.14 ft
	Q =	recharge	= 1.25 ft/yr
	n <sub>e</sub> =	effective porosity	= 0.25
	V =	groundwater flow rate = hydraulic conductivity x hydraulic gradient/ effective porosity	= 19.9 ft/yr
	B =	Aquifer thickness	= 300 ft

Solving the equation yields an estimated value for the thickness of the aquifer of 9 feet. The hydraulic gradient was determined from the water-table map in Figure 3-8. The width of the flow zone was assumed to be the width of the backyard perpendicular to the estimated groundwater flow direction. The volume rate of flow in the aquifer was then estimated to be 1,072 ft<sup>3</sup>.

**Table 5-2**  
**SUMMERS CALCULATION OF THE CONCENTRATION**  
**OF PCE IN GROUNDWATER**

Component	Value
Area	412 ft <sup>2</sup>
Recharge	1.25 ft/yr
Q <sub>p</sub> = volume of infiltration	515 ft <sup>3</sup> /yr
Hydraulic conductivity	1,460 ft/yr
Hydraulic gradient	0.0034
Aquifer thickness	9 ft
Width of flow zone	24 ft
Q <sub>a</sub> = volume of aquifer flow	1,072 ft <sup>3</sup> /yr
C <sub>p</sub> = concentration in soil water	313 μg/l
C <sub>gw</sub> = concentration in groundwater	102 μg/l

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The concentration of PCE in the soil at Gentle Cleaners is variable, as shown in Figures 4-1 and 4-2. To perform the modeling it was necessary to derive a single concentration that was representative of PCE at the facility. The close support laboratory data were used to derive this value, because they were considered to be more representative of the actual values at the facility than were those obtained in the fixed laboratory; concentrations of VOCs obtained from fixed-laboratory analyses are typically less than the concentrations obtained from split samples analyzed in field laboratories because of the loss of VOCs during shipping and handling.

The average PCE concentration at Gentle Cleaners was calculated. The two values that exceeded the field GC analytical limits were not included in the calculation because they were considerably higher than any other concentrations detected at the facility and may not be representative. Taking the average of the remaining concentrations gave a value of about 160  $\mu\text{g}/\text{kg}$ , which was rounded to 200  $\mu\text{g}/\text{kg}$  for modeling purposes. For concentrations less than the detection limit of 2  $\mu\text{g}/\text{kg}$ , a value of 1  $\mu\text{g}/\text{kg}$ , or one-half of the detection limit, was used.

If the two values that exceeded field equipment limits were included in calculating the average, the representative concentration would have been much higher than 200  $\mu\text{g}/\text{kg}$ . Had the two values that exceeded field equipment limits been included and a median calculated, a value of between 5 and 28  $\mu\text{g}/\text{kg}$  would be determined. This value is considered to be too low given that so many other values exceed this range by significant amounts. The implications of these other options for concentration modeling are discussed at the end of this section.

The  $K_d$  can be estimated as the product of the fraction of organic carbon ( $f_{oc}$ ) and the normalized partition coefficient ( $K_{oc}$ ). Values of total organic carbon obtained from samples at the facility ranged from 1,490 to 21,800 mg/kg, with a median value of 2,450 mg/kg. This translates to an  $f_{oc}$  of 0.00245. The  $K_{oc}$  for PCE varies from 209 to 363 ml/gm (Montgomery and Welkom, 1990), with a median of 263 ml/gm. The  $K_d$  is then estimated by  $0.00245 \times 263 \text{ ml/gm} = 0.64 \text{ ml/gm}$ . Then using the relationship  $K_d = C_s/C_p$ ,  $C_p$  is estimated to be 313  $\mu\text{g}/\text{l}$ . The derived values were substituted into the Summers model and a value of 102  $\mu\text{g}/\text{l}$  of PCE in the groundwater immediately below Gentle Cleaners was estimated. The upgradient concentration ( $C_a$ ) was assumed to be zero.

The value of 313  $\mu\text{g}/\text{l}$  of PCE in soil water estimated by this calculation is approximately the concentration of PCE (330  $\mu\text{g}/\text{l}$ ) in the water observed to be entering the GKM well from fractures above the water level in the well. The possibility exists that the contaminated soil water from Gentle Cleaners is entering a fracture system above the water table and is flowing into the GKM well. If this is the case, the 313  $\mu\text{g}/\text{l}$  in the soil water beneath Gentle Cleaners is not available to contaminate the groundwater beneath the cleaners to a value on the order of 102  $\mu\text{g}/\text{l}$ . It is likely that contamination is being transported by both modes.

There are several sources of uncertainty in these calculations. One is how representative the estimated average of the concentration of residual PCE in the soil might be. The value of 200 ug/kg that was used in the calculation provided an estimate of PCE concentration in the soil water that is consistent with the concentration entering the upper part of the GKM well. However, the residual soil concentration could be higher, resulting in higher estimated concentrations in the soil water. This could provide a higher concentration in the groundwater underlying the facility and still provide a concentration to a shallow fracture system that is consistent with that entering the upper part of the GKM well.

Another source of uncertainty is the estimate of the thickness of the aquifer in which the contaminated soil water is mixed. The calculation includes several estimated values. In addition, the calculation assumes that the soil water is mixed throughout the full thickness and extent of the mixing zone. In reality, the convective flow of the aquifer with presumably clean water from upgradient would restrict the mixing to a smaller part of the mixing zone, resulting in a higher overall concentration in the groundwater.

#### **5.5.4 Summary and Conclusions**

On the basis of the calculations using the Summers model, residual PCE contamination at Gentle Cleaners of approximately 200  $\mu\text{g}/\text{kg}$  will result in the contamination of the underlying aquifer to a concentration on the order of 102  $\mu\text{g}/\text{l}$ .

Similar or lower levels of PCE contamination were detected in the soil at Parkside Apartments. Therefore, similar or lower levels of groundwater contamination would be expected from this facility because the dimensions of the facility are similar. Little or no contamination would be expected from the residual soil contamination at Granite Knitting Mills because of the asphalt cover over most of the facility.

There is, of course, considerable uncertainty in the groundwater concentration estimated from this model. Besides the uncertainty associated with the assumptions of the model, there is the uncertainty in the hydraulic parameters such as recharge and hydraulic conductivity used in the model. Therefore, this application of the Summers model represents only a screening approach that indicates the approximate level of groundwater contamination that might be derived from the residual soil contamination.

### **5.6 Summary**

All of the contaminants that were detected have similar fate and transport properties: relatively high water solubilities, high Henry's Law Constants, and low  $K_{oc}$ , which reflect their mobility in soil and water. Concentrations of the VOCs in soil are expected to fall over time, as the contaminants volatilize and move downward. The highest degrees of contamination are in the subsurface soil, generally at the greater depths. These contaminated soils have been and probably continue to be the sources of the groundwater contamination. Contaminants in the groundwater may be anaerobically biologically degrading.

**Section 6**

**Baseline Human Health Risk Evaluation**

AR300781

## Section 6

# Baseline Human Health Risk Evaluation

### 6.1 Introduction

This chapter presents the results of the baseline risk assessment for the North Penn Area I site. The purpose of the baseline risk assessment is to characterize the potential human health and environmental risks from the site. The results of the baseline risk assessment are used to help determine whether remediation is necessary, to help provide justification for performing remedial action, and to assist in determining what exposure pathways need to be remediated.

A baseline risk assessment is an evaluation of the potential threats to public health from the site in the absence of any remedial action (i.e., the no-action alternative). The baseline risk assessment identifies contaminants of potential concern, characterizes their toxicity, identifies the potential exposure pathways, and characterizes the risks to receptors, including a discussion of uncertainty. The baseline risk assessment addresses potential risks from the site under current and future land uses assuming that no corrective actions occur and no restrictions are placed on future use of the land.

### 6.2 Scope

The soils at the site have been separated into three distinct areas, specifically Gentle Cleaners, Parkside Apartments, and Granite Knitting Mills. The groundwater will be treated the same for the entire site. Only the risk due to VOCs will be quantified because there are few or no data on any other potential contaminants, and the primary contaminants identified at the site are VOCs.

### 6.3 Overview of Risk Assessment Process

The risk assessment consists of four major components:

- Identification of chemicals of potential concern
- Toxicity assessment
- Exposure assessment
- Risk characterization

The first step in the risk assessment is to evaluate the data collected as part of the RI, identify contaminants of potential concern, and summarize the data for use in the exposure assessment. This process begins with a review of the contaminants detected at the site. Then, to focus subsequent efforts in the risk assessment, contaminants that are of potential concern to the public health are identified.

The toxicity assessment identifies the types of hazards or health effects associated with exposure to the contaminants of potential concern. The quantitative expressions of dose-response relationships, as well as toxicity values, also are presented.

The exposure assessment identifies potential pathways by which exposure can occur through the use of exposure scenarios. It characterizes the potentially exposed populations and the frequency and duration of the potential exposures.

The risk characterization addresses the potential for adverse effects for each exposure scenario identified during the exposure assessment by integrating the information developed during the toxicity and exposure assessments.

This risk assessment also qualitatively evaluates the sensitivity of the results with regard to some of the assumptions made. The uncertainties associated with the results also are discussed qualitatively.

#### **6.4 EPA Guidance for Risk Assessment**

This risk assessment was performed in accordance with the following guidance and advisories:

- *EPA Risk Assessment Guidelines* (EPA, 1986a,b,c).
- *Exposure Factors Handbook* (EPA, 1989a)
- *Risk Assessment Guidance for Superfund, Volume I, Human Health Evaluation Manual, Part A. Interim Final.* (EPA, 1989b).
- *Human Health Evaluation Manual, Supplemental Guidance: Standard Default Exposure Factors* (EPA, 1991)

#### **6.5 Data Evaluation**

The analytical data collected for the site underwent data validation procedures as described in the North Penn Area I Work Plan. After data validation, the data were reviewed to eliminate results that could represent contamination of samples in the laboratory or in the field or that failed to meet quality control guidelines (e.g., insufficient surrogate spike recovery). Data that met data validation requirements, including estimated results (data with a "J" qualifier) were used in this assessment.



The criteria used in selecting the contaminants of potential concern are as follows:

1. Frequency of Detection: If a contaminant was detected in fewer than 5 percent of the samples, it was dropped as a contaminant of potential concern.
2. Presence in Blanks: Acetone, 2-butanone, methylene chloride, toluene, and the phthalate esters are considered to be common laboratory contaminants. When blanks contained detectable amounts of these contaminants, the sample results were considered as positive results only if the concentrations in the sample exceeded 10 times the maximum amount detected in any blank (EPA, 1989b). If the concentration of any of these common laboratory contaminants in a sample was less than 10 times the maximum blank concentration, it was concluded that the chemical was not present in the sample. For other than these common laboratory contaminants, sample results were considered as positive only if the concentration of the chemical in the sample exceeded five times the maximum amount detected in any blank sample (EPA, 1989b).

Table 6-1 shows the list of contaminants of potential concern that remain after these criteria were applied to the summary of compounds detected in soil and groundwater.

After the contaminants of potential concern were selected, the next step was to develop summary statistics for use in the exposure assessment. The frequency of detection, maximum, average, standard deviation, and 95 percent upper confidence level (calculated using the arithmetic mean) were calculated for the contaminants of potential concern. For samples with nondetects, a value of half the quantitation limit for the sample was assigned for use in the statistical calculations (EPA, 1989b). When the 95 percent upper confidence level is higher than the maximum value for a contaminant, the maximum value was used for the exposure calculations (EPA, 1989b). Tables 6-2 and 6-3 present the summary statistical information for volatiles in subsurface soils and groundwater.

## **6.6 Toxicity Assessment**

### **6.6.1 Definition**

Toxicity assessment has two general steps. The first step, hazard identification, is the process of determining what adverse health effects, if any, could result from exposure to a particular chemical. The second step, dose-response evaluation, quantitatively examines the relationship between the level of exposure and the incidence of adverse health effects in an exposed population.

**Table 6-1**  
**CONTAMINANTS OF CONCERN**

Acetone  
Benzene  
Bromodichloromethane  
Carbon disulfide  
Chloroform  
Chloromethane  
1,1-Dichloroethane  
Cis/trans-1,2-Dichloroethene  
Ethylbenzene  
Methylene chloride  
Tetrachloroethene  
Toluene  
1,1,1-Trichloroethane  
Trichloroethene

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Table 6-2  
VOCS IN SUBSURFACE SOIL

Sample Matrix:	Soil											
Sample Analysis:	Volatiles Organics	Chemical Name	Concentration Units	Number of Samples	Number Detected	Detected Frequency	Detected Minimum	Detected Maximum	Detected Average	Standard Deviation	Geometric Mean	95% Confidence Limit
<b>PARKSIDE APARTMENTS</b>												
		1,2-Dichloroethene (total)	MG/KG	6	1	0.1667	0.004	0.004	0.004	0.000	0.006	0.01
		Acetone	MG/KG	6	5	0.8333	0.008	0.018	0.012	0.003	0.010	0.02
		Methylene Chloride	MG/KG	6	6	1.0000	0.013	0.048	0.028	0.013	0.024	0.06
		Tetrachloroethene	MG/KG	6	3	0.5000	0.005	0.099	0.053	0.038	0.013	0.92
		Trichloroethene	MG/KG	6	1	0.1667	0.029	0.029	0.029	0.000	0.008	0.02
<b>GRANITE KNITTING MILLS</b>												
		2-Butanone	MG/KG	5	1	0.2000	0.017	0.017	0.017	0.000	0.007	0.02
		Acetone	MG/KG	5	5	1.0000	0.012	0.031	0.021	0.007	0.019	0.04
		Methylene Chloride	MG/KG	5	5	1.0000	0.011	0.033	0.016	0.008	0.015	0.03
		Tetrachloroethene	MG/KG	5	3	0.6000	0.006	6.9	1.73	2.985	0.020	2,691,771.27
		Toluene	MG/KG	5	1	0.2000	0.012	0.012	0.012	0.000	0.007	0.01
<b>GENTLE CLEANERS</b>												
		Acetone	MG/KG	4	4	1.0000	0.007	0.048	0.028	0.018	0.021	2.85
		Methylene Chloride	MG/KG	4	4	1.0000	0.012	0.018	0.015	0.002	0.015	0.02
		Tetrachloroethene	MG/KG	4	2	0.5000	0.100	300.000	100.097	141.353	0.180	****

Rejected observations are not included in any calculations. Detected frequency = detected count/total count.

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Table 6-3  
VOCs in Groundwater

EDMS CHEMICAL SUMMARY STATISTICS BY SAMPLE MATRIX  
/VOLATILE ORGANICS - CURRENT GROUPING  
ITCs not included  
SAMPLE MATRIX: GROUNDWATER  
SAMPLE ANALYSIS: VOLATILE ORGANICS

Chem Code	Chemical Name	Conc Units	Total Count	Detected Count	Detected Frequency	Detected Minimum	Detected Maximum	Detected Average	Standard Deviation	Geometric Mean	95% Confidence Limit
111	1,1,1-TRICHLOROETHANE	UG/L	13	1	0.0769	0.100	0.100	0.100	0.000	0.442	0.64
11E	1,1,2-TETRACHLOROETHANE	UG/L	13	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
112	1,1,2-TRICHLOROETHANE	UG/L	13	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
11A	1,1-DICHLOROETHANE	UG/L	13	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
1DE	1,1-DICHLOROETHENE	UG/L	13	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
D3C	1,2-DIBROMO-3-CHLOROPROPANE	UG/L	13	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
12E	1,2-DIBROMETHANE	UG/L	13	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
12B	1,2-DICHLOROBENZENE	UG/L	13	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
12A	1,2-DICHLOROETHANE	UG/L	13	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
12P	1,2-DICHLOROPROPANE	UG/L	13	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
13B	1,3-DICHLOROBENZENE	UG/L	13	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
14B	1,4-DICHLOROBENZENE	UG/L	13	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
2BU	2-BUTANONE	UG/L	13	0	0.0000	0.000	0.000	0.000	0.000	2.499	1.00
2HX	2-HEXANONE	UG/L	13	0	0.0000	0.000	0.000	0.000	0.000	2.499	1.00
4M2	4-METHYL-2-PENTANONE	UG/L	13	0	0.0000	0.000	0.000	0.000	0.000	2.499	1.00
ACT	ACETONE	UG/L	13	0	0.0000	0.000	0.000	0.000	0.000	2.499	1.00
BEN	BENZENE	UG/L	13	2	0.1538	0.030	0.050	0.040	0.010	0.337	1.18
BCH	BROMOCHLOROMETHANE	UG/L	13	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
BDM	BROMODICHLOROMETHANE	UG/L	13	1	0.0769	0.100	0.100	0.100	0.000	0.442	0.64
BFM	BROMOFORM	UG/L	13	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
BRM	BROMOMETHANE	UG/L	13	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
CDS	CARBON DISULFIDE	UG/L	13	1	0.0769	0.100	0.100	0.100	0.000	0.442	0.64
CCL	CARBON TETRACHLORIDE	UG/L	13	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
CBN	CHLOROBENZENE	UG/L	13	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
CET	CHLOROETHANE	UG/L	13	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
CEM	CHLOROFORM	UG/L	13	1	0.0769	0.200	0.200	0.200	0.000	0.466	0.55
CHE	CHLOROMETHANE	UG/L	13	11	0.8462	0.100	2.000	0.600	0.522	0.440	1.06
CPD	CIS-1,3-DICHLOROPROPENE	UG/L	13	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
CTD	CIS/TRANS-1,2-DICHLOROETHENE	UG/L	13	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
DBM	DIBROMOCHLOROMETHANE	UG/L	13	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
EDD	ETHYLBENZENE	UG/L	13	9	0.6923	0.020	0.070	0.031	0.017	0.067	0.91
MCL	METHYLENE CHLORIDE	UG/L	13	0	0.0000	0.000	0.000	0.000	0.000	1.000	1.00
STY	STYRENE	UG/L	13	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
PTD	TETRACHLOROETHENE	UG/L	13	3	0.2308	0.050	3.000	1.050	1.379	0.425	1.42
TOL	TOLUENE	UG/L	13	5	0.3846	0.030	0.200	0.126	0.056	0.273	0.85

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REJECTED OBSERVATIONS ARE NOT INCLUDED IN ANY CALCULATIONS. DETECTED FREQUENCY = DETECTED COUNT/TOTAL COUNT.

Table 6-3  
(Continued)

DMS CHEMICAL SUMMARY STATISTICS BY SAMPLE MATRIX  
VOLATILE ORGANICS - CURRENT GROUPING  
ICs not included  
SAMPLE MATRIX: GROUNDWATER  
ANALYSIS: VOLATILE ORGANICS

Chem Code	Chemical Name	Conc Units	Total Count	Detected Count	Detected Frequency	Detected Minimum	Detected Maximum	Detected Average	Standard Deviation	Geometric Mean	95% Confidence Limit
11E	TRANS-1,2-DICHLOROETHENE	UG/L	13	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
113	TRANS-1,3-DICHLOROPROPENE	UG/L	13	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
1CE	TRICHLOROETHENE	UG/L	13	1	0.0769	0.300	0.300	0.300	0.000	0.481	0.52
1JC	VINYL CHLORIDE	UG/L	13	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
1XY	XYLENE (TOTAL)	UG/L	13	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00

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Table 6-3  
(Continued)

MS SUMMARY STATISTICS BY SAMPLE MATRIX  
VOLATILE ORGANICS - FUTURE GROUPING  
ICs not included  
SAMPLE MATRIX: GROUNDWATER  
ANALYSIS: VOLATILE ORGANICS

Chemical Name	Conc Units	Total Count	Detected Count	Detected Frequency	Detected Minimum	Detected Maximum	Detected Average	Standard Deviation	Geometric Mean	95% Confidence Limit
1,1,1-TRICHLOROETHANE	UG/L	5	2	0.4000	0.000	0.800	0.900	0.050	0.618	0.91
1,1,1,2-TETRACHLOROETHANE	UG/L	5	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
1,1,2-TRICHLOROETHANE	UG/L	5	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
1,1,1-DICHLOROETHANE	UG/L	5	2	0.4000	0.300	0.400	0.350	0.050	0.432	0.57
1,1-DICHLOROETHANE	UG/L	5	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
1,2-DIBROMO-3-CHLOROPROPANE	UG/L	5	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
1,2-DIBROMOETHANE	UG/L	5	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
1,2-DICHLOROBENZENE	UG/L	5	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
1,2-DICHLOROETHANE	UG/L	5	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
1,2-DICHLOROPROPANE	UG/L	5	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
1,3-DICHLOROBENZENE	UG/L	5	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
1,4-DICHLOROBENZENE	UG/L	5	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
2-BUTANONE	UG/L	5	0	0.0000	0.000	0.000	0.000	0.000	2.499	1.00
2-HEXANONE	UG/L	5	0	0.0000	0.000	0.000	0.000	0.000	2.499	1.00
4-METHYL-2-PENTANONE	UG/L	5	0	0.0000	0.000	0.000	0.000	0.000	2.499	1.00
ACETONE	UG/L	5	0	0.0000	0.000	0.000	0.000	0.000	2.499	1.00
BENZENE	UG/L	5	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
BROMOCHLOROMETHANE	UG/L	5	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
BROMODICHLOROMETHANE	UG/L	5	2	0.4000	0.100	0.100	0.100	0.000	0.263	2.69
BROMOFORM	UG/L	5	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
BROMOMETHANE	UG/L	5	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
CARBON DISULFIDE	UG/L	5	2	0.4000	0.100	0.100	0.400	0.300	0.388	2.39
CARBON TETRACHLORIDE	UG/L	5	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
CHLOROBENZENE	UG/L	5	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
CHLOROETHANE	UG/L	5	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
CHLOROFORM	UG/L	5	1	0.2000	0.200	0.200	0.200	0.000	0.416	0.78
CHLOROMETHANE	UG/L	5	1	0.2000	0.200	0.200	0.200	0.000	0.416	0.78
CIS-1,3-DICHLOROPROPENE	UG/L	5	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
CIS/TRANS-1,2-DICHLOROETHENE	UG/L	5	3	0.6000	0.300	1.000	0.767	0.330	0.596	1.48
DIBROMOCHLOROMETHANE	UG/L	5	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
ETHYLBENZENE	UG/L	5	1	0.2000	0.030	0.030	0.030	0.000	0.285	28.19
METHYLENE CHLORIDE	UG/L	5	0	0.0000	0.000	0.000	0.000	0.000	1.000	1.00
STYRENE	UG/L	5	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
TETRACHLOROETHENE	UG/L	5	5	1.0000	0.300	5.000	2.460	1.663	1.684	70.04
TOLUENE	UG/L	5	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00

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REJECTED OBSERVATIONS ARE NOT INCLUDED IN ANY CALCULATIONS. DETECTED FREQUENCY = DETECTED COUNT/TOTAL COUNT.

Table 6-3  
(Continued)

DMS CHEMICAL SUMMARY STATISTICS BY SAMPLE MATRIX  
VOLATILE ORGANICS - FUTURE GROUPING  
ICs not included  
SAMPLE MATRIX: GROUNDWATER  
SAMPLE ANALYSIS: VOLATILE ORGANICS

Chem Code	Chemical Name	Conc Units	Total Count	Detected Count	Detected Frequency	Detected Minimum	Detected Maximum	Detected Average	Standard Deviation	Geometric Mean	95% Confidence Limit
ITE	TRANS-1,2-DICHLOROETHENE	UG/L	5	1	0.2000	0.200	0.200	0.200	0.000	0.416	0.78
IT3	TRANS-1,3-DICHLOROPROPENE	UG/L	5	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
ICE	TRICHLOROETHENE	UG/L	5	4	0.8000	0.200	0.700	0.425	0.179	0.407	0.87
VC	VINYL CHLORIDE	UG/L	5	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00
XY	XYLENE (TOTAL)	UG/L	5	0	0.0000	0.000	0.000	0.000	0.000	0.500	1.00

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For the purpose of this risk assessment, human health effects are divided into two categories: carcinogenic and noncarcinogenic effects. Consequently, human health risks are evaluated in this assessment in terms of carcinogenic and noncarcinogenic risks. Chemicals with carcinogenic risk frequently have noncarcinogenic effects, too.

The requirements of a toxicity assessment include:

- Collection of toxicity information for the chemicals of potential concern
- Identification of the exposure duration (subchronic, chronic) for which toxicity values are derived
- Selection of the appropriate toxicity values for noncancerous effects
- Selection of the appropriate toxicity values for cancerous effects
- A summary of the toxicity information for the chemicals of potential concern

### **6.6.2 Collection of Toxicity Information**

Toxicity information regarding the chemicals of potential concern is available from the toxicology literature and databases maintained by the EPA. Epidemiological studies that demonstrate a significant correlation between the level of exposure to a chemical and the development of an adverse health effect provide the most convincing evidence to support an assessment of the risk to human health. For example, epidemiological studies of human populations exposed to arsenic in drinking water form the basis of the arsenic risk assessment conducted by the EPA to develop numerical estimates of the cancer and noncancer toxicity. However, except for a few chemicals, epidemiological studies are not available or are inadequate as the sole basis for a risk assessment. Toxicology studies in animals are, therefore, used to support or replace inadequate or unavailable epidemiological studies.

Studies in animals are of the greatest value when the effects observed also have been observed in humans and when similar effects are observed in more than one species and by more than one route of exposure. Some VOCs, for example, when inhaled or ingested, have been observed to cause neurological effects in both humans and animals. Other types of toxicology studies, including pharmacokinetic, mutagenicity, and genotoxicity studies, also can provide supporting information in evaluating a chemical's toxicity.



Pharmacokinetic studies can reveal the degree to which observations of toxicity in humans and animals are comparable. Toxicology studies involving animals are particularly valuable when the animal's metabolism is similar to that of humans. Genotoxicity and mutagenicity studies can provide information regarding the carcinogenicity of a chemical and the mechanism by which cancer may develop.

For many chemicals, chemical-specific toxicity values have been developed on the basis of the toxicology literature for use in the preparation of human health risk assessments. Separate toxicity values are developed for noncancerous effects and cancer because these two types of toxicity are thought to develop by different mechanisms.

### **6.6.3 Definition of a Reference Dose**

A reference dose, or RfD, is the toxicity value used most often in baseline risk assessments to evaluate noncancerous effects that result from chemical exposures. A chronic RfD is defined as an estimate, with uncertainty spanning an order of magnitude or more, of a daily exposure level for the human population, including sensitive subpopulations, that is likely to be without an appreciable risk of deleterious effects during a lifetime. Chronic RfDs are specifically developed to be protective for long-term exposure to a compound. As a guide, chronic RfDs generally should be used to evaluate the potential noncancerous effects associated with exposure periods between 7 years (approximately 10 percent of a human lifetime) and a lifetime. Many chronic RfDs have been reviewed and verified by a EPA RfD work group and entered into the EPA Integrated Risk Information System (IRIS). Chronic RfDs that have yet to be reviewed and verified are entered on an interim basis in the EPA Health Effects Assessment Summary Table (HEAST).

### **6.6.4 Definition of a Slope Factor**

The dose-response relationship for carcinogens is expressed as a cancer slope factor. Generally, the slope factor is a plausible upper-bound estimate of the probability of an occurrence of cancer per unit intake of a chemical over a lifetime. The slope factor is usually the upper 95 percent confidence limit of the slope of the dose-response curve and is expressed in units of risk per milligram of chemical per kilogram of body weight per day  $(\text{mg}/\text{kg}/\text{day})^{-1}$ .

For practical reasons, risk at low exposure levels is difficult to measure directly either by animal experiments or epidemiologic studies. The development of a slope factor generally entails applying a model to the available data set. It is used to extrapolate from the relatively high doses administered to experimental animals (or exposures reported in epidemiologic studies) to lower exposure levels expected for human contact in the environment. A basic assumption is that, if a carcinogenic response occurs at the dose levels used in the study, a response will occur at all lower doses.

If the extrapolation model selected is the linearized multistage model, the slope factor is also known as the  $q_1^*$ . Occasionally, slope factors are based on human epidemiologic data (e.g., arsenic and benzene) and are based on the "best" estimate instead of upper 95 percent confidence limits. Use of slope factors assumes that cancer risk is probabilistic and any degree of exposure leads to some degree of risk.

### **6.6.5 Carcinogen Classification**

The EPA has developed a carcinogen classification system that uses a weight-of-evidence approach to classify the likelihood of a chemical being a human carcinogen. Information considered in developing the classification includes human studies of the association between cancer incidence and exposure as well as long-term animal studies under controlled laboratory conditions. The EPA classifies chemicals as:

- A—Human carcinogen
- B1—Probable human carcinogen; limited human data are available
- B2—Probable human carcinogen; sufficient evidence in animals and inadequate or no evidence in humans
- C—Possible human carcinogen
- D—Not classifiable as to human carcinogenicity
- E—Evidence of noncarcinogenicity for humans

### **6.6.6 Sources of Toxicity Values**

Toxicity values are available for many chemicals from EPA databases, including IRIS (1992) and HEAST (1992). IRIS is the primary source for RfD, weight-of-evidence, and slope factor values. IRIS is routinely updated by the EPA and contains toxicity information files and chemical-specific toxicity values that have undergone EPA review and verification. HEAST contains interim RfD, weight-of-evidence, and slope factor values before they are reviewed and verified for entry on IRIS. HEAST is updated annually. EPA Region III has compiled RfDs and slope factor values for nearly 600 chemicals (Roy Smith, "Risk-Based Concentration Table, Third Quarter 1992").

Summary toxicity values for carcinogens are presented in Table 6-4 and for noncarcinogens in Table 6-5. The toxicity values for trichloroethene are based on the levels prior to its withdrawal from IRIS.

**Table 6-4  
CANCER TOXICITY VALUES<sup>a</sup>**

<b>Chemical</b>	<b>Carcinogen Weight-of- Evidence Classification</b>	<b>Oral Slope Factor (kg-day/mg)</b>	<b>Inhalation Slope Factor (kg-day/mg)</b>
Benzene	A	0.029	0.0291
Bromodichloromethane	B2	0.13	NA <sup>b</sup>
Chloroform	B2	0.0061	0.0805
Chloromethane	C	0.013	0.0063
Methylene Chloride	B2	0.0075	0.00165
Tetrachloroethene	B2	0.052	0.00203
Trichloroethene <sup>c</sup>	B2	0.011	0.006

<sup>a</sup>Sources:

EPA Integrated Risk Information System (IRIS), updated July 1992.  
EPA Health Effects Assessment Summary Tables (HEAST), annual  
FY 1991.

EPA Office of Health and Environmental Assessment, ECAO-Cincinnati.  
(The toxicity values from these sources are presented in EPA-Region III  
"Risk-Based Concentration Table, Third Quarter 1992," Roy Smith, July  
1992).

<sup>b</sup>The oral slope factor will be used for the inhalation risk characterization  
calculations.

<sup>c</sup>The toxicity values for trichloroethene used here are as agreed to with EPA  
Region III.

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**Table 6-5  
NONCANCER TOXICITY VALUES<sup>a</sup>**

Chemical	Oral RfD (mg/kg-day)	Inhalation RfD (mg/kg-day)
Acetone	0.10	NA <sup>b</sup>
Bromodichloromethane	0.02	NA <sup>b</sup>
Carbon Disulfide	0.1	0.00286
Chloroform	0.01	NA <sup>b</sup>
1,1-Dichloroethane	0.1	0.143
Cis-1,2-Dichloroethene	0.01	NA <sup>b</sup>
Ethylbenzene	0.1	0.286
Methylene Chloride	0.06	0.857
Tetrachloroethene	0.01	NA <sup>b</sup>
Toluene	0.2	0.114
1,1,1-Trichloroethane	0.09	0.286
Trichloroethene <sup>c</sup>	0.006	NA <sup>b</sup>

<sup>a</sup>Sources:

EPA Integrated Risk Information System (IRIS), updated July 1992  
 EPA Health Effects Assessment Summary Tables (HEAST), annual  
 FY 1991.

EPA Office of Health and Environmental Assessment, ECAO-Cincinnati  
 (The toxicity values from these sources are presented in EPA-Region III  
 "Risk-Based Concentration Table, Third Quarter 1992," Ray Smith, July  
 1992)

<sup>b</sup>The oral RfD will be used for the inhalation risk characterization calculations.

<sup>c</sup>The toxicity value for trichloroethene used here is as agreed to with EPA  
 Region III.

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### 6.6.7 Toxicity Profiles

Summary toxicity profiles are presented for a select number of chemicals of potential concern. The summary profiles for the chemicals of potential concern are presented below. The profiles describe four categories of potential toxic effects: acute toxicity, chronic toxicity, carcinogenicity, and other effects. Although toxicity profiles are not provided for all chemicals detected at the site, the exclusion of a chemical is not meant to imply that exposure to that chemical is without effect.

#### *Acetone*

**Acute Toxicity Summary.** Exposure to acetone can cause irritation of the eyes, skin, and respiratory tract. Depression of the central nervous system (CNS) and narcosis can occur following inhalation of high concentrations of acetone; dryness of the mouth and throat, dizziness, nausea, incoordination, loss of speech, and even coma have been described in some cases of workers exposed occupationally to acetone.

**Chronic Toxicity Summary.** In a study reported by EPA (IRIS 3/1/88), kidney damage was observed in albino rats administered acetone by oral gavage. Kidney damage and metabolic changes have been noted in humans who ingested acetone (Sax 1989).

**Cancer Potential.** None is indicated.

**Other.** Acetone can potentiate the toxicity of other chemicals, particularly solvents. The hepatotoxicity of carbon tetrachloride can be increased greatly in the presence of acetone.

#### *Benzene*

**Acute Toxicity Summary.** Acute exposures (inhalation) to high levels of benzene may lead to depression of the CNS, unconsciousness, and death or may cause fatal cardiac arrhythmias.

**Chronic Toxicity Summary.** The major toxic effect is hematopoietic toxicity (affects formation of blood). Chronic exposure of workers to low levels has been associated with blood disorders, such as leukemia and aplastic anemia (depression of all three cell types of the blood in the absence of functioning marrow).

**Cancer Potential.** Sufficient evidence that benzene is a human and animal carcinogen; classified by EPA as an "A" human carcinogen. A strong correlation exists between exposure to benzene by inhalation and leukemia.

**Other.** Chromosomal aberrations in bone marrow and blood have been reported in experimental animals and some workers.

### ***Bromodichloromethane***

**Acute Toxicity Summary.** Acute effects in laboratory animals include sedation, anesthesia, fatty infiltration of the liver, and hemorrhage in kidney, adrenals, lungs, and brain.

**Chronic Toxicity Summary.** Long-term studies in mice show reduced body weight, centrilobular degeneration of the liver, depressed hepatic reticuloendothelial system, tubular cell hyperplasia and cytomegaly of the kidney, and immunological suppression.

**Cancer Potential.** None is indicated.

**Other.** Other effects occur from chlorination of natural organic precursors in raw water. Fetotoxicity is possible.

### ***Carbon Disulfide.***

**Acute Toxicity Summary.** Acute inhalation and oral exposures to carbon disulfide affect the nervous system, the cardiovascular system, and the liver (ATSDR 1991).

**Chronic Toxicity Summary.** Chronic occupational exposures have shown that the primary targets following inhalation of carbon disulfide are the nervous system, the cardiovascular system, the liver, and the eye (ATSDR 1991).

**Cancer Potential.** None is indicated.

### ***Chloroform***

**Acute Toxicity Summary.** Chloroform is an anesthetic that depresses the CNS. Fatalities in humans may be rapid, resulting from cardiac arrest (apparently sensitization to epinephrine) or delayed with kidney and liver damage. Respiratory depression, coma, and liver and kidney damage are among the symptoms of exposure to chloroform. In laboratory animals, acute toxicity depends on species, strain, sex, and age; liver damage may be cause of death in rats and mice after acute exposure.

**Chronic Toxicity Summary.** Kidney damage (renal tubular necrosis) can occur in mice, and kidney and liver damage can occur in rats, rabbits, dogs, and guinea pigs exposed by inhalation.

**Cancer Potential.** Chloroform is carcinogenic in mice (hepatomas, hepatocellular carcinomas), male rats (malignant kidney tumors), and female rats (thyroid tumors).

**Other.** It is fetotoxic in rats and rabbits.

## *Chloromethane*

**Acute Toxicity Summary.** Adverse effects observed in humans during or after inhalation exposure to chloromethane include central nervous system effects, such as nausea, vomiting, confusion, staggering, slurred speech, convulsions, coma, and death and cardiovascular effects, such as tachycardia and decreased blood pressure. Liver damage has been observed in animals exposed for several days up to one year. Kidney damage, including tubular degeneration and necrosis, has been observed in animals.

**Chronic Toxicity Summary.** Anecdotal evidence of liver damage after chronic occupational exposure has been reported.

Chronic exposure of animals (rats and mice) to chloromethane resulted in increased mortality rates, liver degeneration, kidney hyperplasia, splenic atrophy (mice only), neurotoxicity (cerebellar lesions in mice), and testicular atrophy.

**Carcinogenicity.** No statistical evidence of an increase in cancer mortality was found in an epidemiological study of workers exposed to chloromethane in a butyl rubber manufacturing plant.

Male mice exposed to high concentrations of chloromethane for up to 12 months developed renal tumors. No evidence of carcinogenicity was observed in female mice or in male or female rats from the same study.

**Other.** Reproductive and developmental toxicity, including decreased fertility, testicular lesions and degeneration, fetal heart defects, and post-implantation loss have been observed in studies of rats and mice. Testicular degeneration was not observed in cats exposed for 3 days or in dogs exposed for 90 days to chloromethane.

## *1,1-Dichloroethane*

**Acute Toxicity Summary.** Central nervous system depression may occur when 1,1-dichloroethane is inhaled at high concentrations. It also is irritating to skin.

**Chronic Toxicity Summary.** Data are limited.

**Cancer Potential.** None is indicated.

### *cis-1,2-Dichloroethene*

**Acute Toxicity Summary.** An anesthetic at high concentrations, *cis*-1,2-DCE appears half as potent as *trans*-isomer in depressing CNS; elevated liver enzymes in rats have been reported after one exposure.

**Chronic Toxicity Summary.** Minimal fatty accumulation occurs in the liver of rats chronically exposed to high doses of *cis*-1,2-DCE in drinking water.

**Cancer Potential.** None is indicated.

### *trans-1,2-Dichloroethene*

**Acute Toxicity Summary.** Inhalation exposure to high levels can cause narcosis and death in rats.

**Chronic Toxicity Summary.** Rats exposed by inhalation exhibited fatty accumulation in liver and infiltration of lungs.

**Cancer Potential.** None is indicated.

### *Ethylbenzene*

**Acute Toxicity Summary.** Ethylbenzene is irritating to eyes, mucous membranes, and skin. It can cause headaches and narcosis.

**Chronic Toxicity Summary.** Data are limited.

**Cancer Potential.** None is indicated.

### *Methylene Chloride*

**Acute Toxicity Summary.** Associated with impairment in function of the central nervous system and liver and kidney effects. In human experimental inhalation studies, methylene chloride decreased visual and auditory functions and impaired psychomotor tasks (ATSDR 1988).

**Chronic Toxicity Summary.** Chronic inhalation exposure to methylene chloride has been associated with mild liver toxicity as shown by cytoplasmic vacuolization, increased fat content, and multinucleated hepatocytes. Histomorphological alterations and fatty changes also were noted in mice and rats following oral exposure (ATSDR 1988).

**Cancer Potential.** EAP has classified methylene chloride as a probable human carcinogen (B2).



## *Tetrachloroethene (Perchloroethylene)*

**Acute Toxicity Summary.** Tetrachloroethene can depress the CNS and cause narcosis. It is irritating to mucous membranes and skin and can cause lung edema. Neurological effects on dry-cleaning workers have been reported.

**Chronic Toxicity Summary.** Chronic exposure may result in pathological changes in liver of laboratory animals. It also may affect the kidney. In humans, inhalation exposure may produce irritation of respiratory tract, nausea, headache, sleeplessness, and abdominal pains. Fatalities have been reported.

**Cancer Potential.** It is carcinogenic in laboratory animals. An increased incidence of cancers among dry-cleaning workers exposed to several solvents has been described.

## *Toluene*

**Acute Toxicity Summary.** Acute inhalation of high concentrations of toluene by experimental animals can result in CNS depression and signs of narcotic effects. Humans exposed by inhalation experimentally, occupationally, or through intentional abuse may exhibit excitation, then CNS depression. Neurotoxic effects include nausea, fatigue, and loss of coordination at low levels of exposure and confusion, ataxia, and weakness at higher levels. In rats, irritation of mucous membranes, loss of coordination, and pulmonary irritation have been observed with subchronic exposure. Toluene is slightly irritating to the skin and eyes of rabbits.

**Chronic Toxicity Summary.** CNS and behavioral effects have been observed in subacute or subchronic studies of rats, mice, dogs, and cats. CNS effects reported in workers include disturbances of memory and thinking, psychomotor skills, visual accuracy, sensorimotor speed, and performance tests. Indications of cerebral and cerebellar dysfunction include tremors, ataxia, and equilibrium disorders. In addition, bizarre behavior and emotional lability may occur. In cases of abuse, changes in liver and kidney function have been observed. Effects on the kidney have been demonstrated in rats and dogs, and effects on liver enzymes and liver weight were observed in subacute studies of rats.

**Cancer Potential.** IARC has determined that toluene is not classifiable as to its carcinogenicity to humans, with inadequate evidence for carcinogenicity in experimental animals and humans. One oral study and one inhalation study in rats were judged inadequate, and skin-painting studies were negative or inadequate. Subsequently, a National Toxicology Program (NTP) inhalation study was performed on rats and mice. No evidence of carcinogenic activity was found.

**Other.** Embryotoxicity has been found in some studies of mice and rats but not in studies of rabbits, and the embryotoxicity generally occurred concurrently with maternal toxicity. In rats, skeletal retardation of offspring has been described. Embryo lethality, reduction of fetal weight, and possible teratogenicity in mice were reported in an abstract. Mutagenicity studies in humans are inconclusive; increased frequency of sister chromatic exchange and chromosomal aberrations in peripheral lymphocytes were observed in one study of workers but not in several other studies. Chromosomal aberrations in rats and micronuclei in mice and rats were induced by toluene; however no morphologic transformation occurred in cultured animal cells. DNA damage was induced in cultured animal cells but not in bacteria.

### ***1,1,1-Trichloroethane***

**Acute Toxicity Summary.** Trichloroethane is a CNS depressant and may impair psychophysiological functions. Human fatalities have been reported following deliberate inhalation or occupational exposures; lung congestion was found.

**Chronic Toxicity Summary.** Exposure by inhalation can produce liver damage in mice and affects drug metabolism in the livers of rats.

**Cancer Potential.** Mutagenic in some in vitro tests.

**Other.** Not determined.

### ***Trichloroethene (TCE)***

**Acute Toxicity Summary.** Exposure to TCE can cause depression of the CNS, including dizziness, headaches, incoordination similar to that induced by alcohol, nausea, vomiting, and unconsciousness.

**Chronic Toxicity Summary.** Long-term inhalation exposure can affect liver and kidneys in animals. In humans, changes in liver enzymes have been associated with TCE exposure.

**Cancer Potential.** Exposure of mice (orally and by inhalation) and rats have produced increases in liver or lung or kidney tumors.

**Other.** "Degreasers flush" has been described in TCE-exposed workers who consume alcohol.

## **6.6.8 Sources of Uncertainty in the Toxicity Assessment**

Assessing the toxicity of the chemicals of potential concern at the site requires simplifying assumptions regarding the likelihood that these chemicals will cause the effects observed in studies of humans and other animals. This simplification can introduce significant uncertainty. The sources of uncertainty in this toxicity assessment are listed in Table 6-6.

## **6.7 Exposure Assessment**

Exposure assessment is the evaluation of the means by which individuals may come into contact with contaminants from the site and the magnitude of exposure that could result from that contact. The assessment may be qualitative or quantitative. A qualitative assessment involves a critical evaluation of the parameters affecting the likelihood and magnitude of an exposure through a given pathway and may result in a qualitative judgment about the importance of the exposure pathway. When quantitative, estimates of exposure are used to calculate human intake of a contaminant, a step necessary for assessing the possible health impact or risk associated with an exposure.

The following section presents the assessment of potential exposure to contaminants detected in surface soils, subsurface soils, and groundwater. It addresses exposures that could occur during both current and potential future uses of the property based on a set of assumptions.

The assessment proceeds in three steps. The first is the definition of current and potential land uses. This was done in an earlier chapter. The second is the identification and screening of exposure pathways that may exist under current and future uses. The third is the presentation of the exposure scenarios and other assumptions used to estimate the magnitude of potential human exposure through each pathway identified in the previous step.

### **6.7.1 Identification and Screening of Exposure Pathways**

An exposure pathway is the route by which a contaminant moves from a source to a point of human exposure. A complete exposure pathway has five components:

- Contaminant source
- Mechanism for contaminant release
- Environmental transport medium
- Exposure point (human location)
- Feasible route of exposure (ingestion, inhalation, dermal contact)

**Table 6-6  
SOURCES OF UNCERTAINTY IN THE TOXICITY ASSESSMENT**

<b>Uncertainty Factor</b>	<b>Effect of Uncertainty</b>	<b>Comment</b>
Use of Critical Toxicity Values	May Over- or Under-Estimate Risk	Not all values represent the same degree of certainty; all are subject to change as new information becomes available
Critical Toxicity Values Derived Primarily from Animal Studies	May Over- or Under-Estimate Risk	Extrapolation from animals to humans may result in errors due to pharmacodynamic and pharmacokinetic differences, target organs, and population variability
Critical Toxicity Values Derived Primarily from High Doses; Most Exposures are at Low Doses	May Over-Estimate Risks	Assumes linearity at low doses; tends to have conservative exposure assumptions

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Exposure cannot occur unless each element of the pathway exists or is reasonably expected to exist, thereby completing the link between the source and the exposed individual.

A number of potential human exposure pathways have been identified at the site. These include:

- Soil—Existing and potential future ingestion and dermal absorption of contaminants from surface and subsurface soil.
- Groundwater—Potential future residential use with ingestion, dermal absorption, and inhalation of volatilized compounds.
- Air—Existing and potential future inhalation of volatiles from contaminated soil.

Figure 6-1 illustrates potential exposure pathways from the release of contaminated soil. Only some of these pathways will be completed (e.g., contaminant uptake by a receptor) for current and future land use.

Table 6-7 presents the exposure pathways addressed in this baseline risk assessment, highlighting which pathways will have a quantitative assessment and which will be assessed qualitatively. Subsurface soil is included as a potential exposure pathway for future residents because soil excavation, such as for the installation of home foundations, could occur. Such excavation could bring subsurface soils to the surface allowing direct exposure to contaminants detected in subsurface soils.

### **6.7.2 Quantification of Exposure**

After the potential receptor populations have been identified and the potential exposure pathways have been determined, the next step is to quantify the intake of the contaminants. Six basic factors are used to calculate intake: (1) exposure frequency, (2) exposure duration, (3) environmental media contact rate, (4) chemical concentration, (5) body weight, and (6) averaging time.

Exposure frequency is an indication of the number of times per day or per year that an individual comes into contact with a contaminant. Exposure duration is the length of time of contact with the contaminated media. The contact rate refers to the rates of inhalation, ingestion, and dermal contact, depending on the route of exposure. For example, the contact rate for ingestion is simply the amount of contaminated media such as water (or food) containing the contaminant of potential concern that an individual ingests in a specific time period, such as per day. The chemical concentration is the concentration of the contaminant in the medium (air, water, etc.) contacting the body. The value for body weight is the average body weight over the

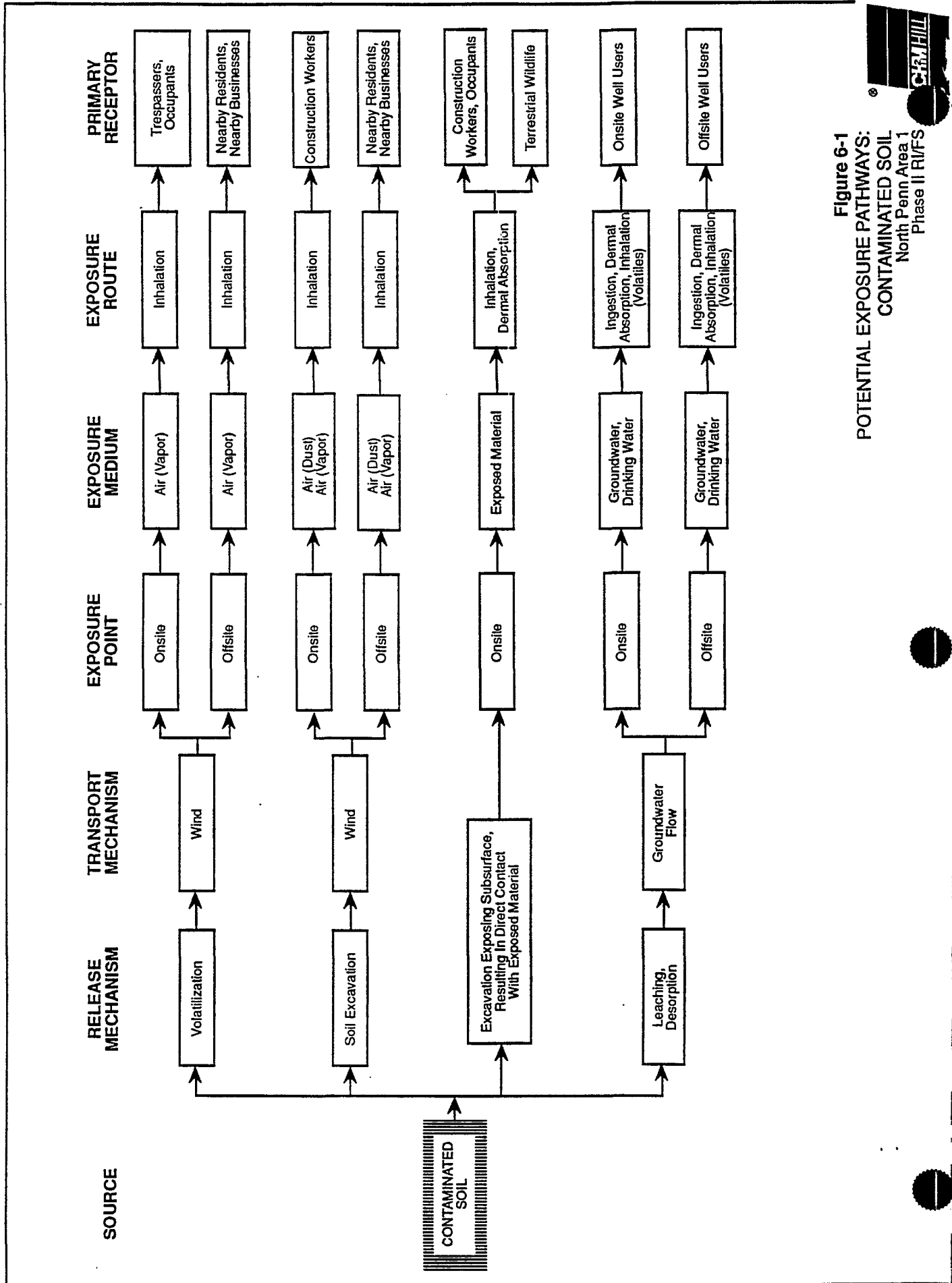


Figure 6-1  
POTENTIAL EXPOSURE PATHWAYS:  
CONTAMINATED SOIL  
North Penn Area 1  
Phase II RI/FS



**Table 6-7  
EXPOSURE PATHWAYS ADDRESSED**

<b>Receptor (Onsite)</b>	<b>Media</b>	<b>Exposure Route</b>	<b>Quantitative Assessment</b>	<b>Qualitative Assessment</b>
<b>Current Land Use</b>				
Residents—Onsite	Surface Soil	Ingestion Dermal Absorption		X X
	Air (outdoors) (indoors)	Inhalation Inhalation		X X
Residents—Offsite	Groundwater	Ingestion Dermal Absorption Inhalation (Indoor Volatile)	X X X	
Workers—Onsite	Surface Soil	Ingestion Dermal Absorption		X X
	Air (outdoors)	Inhalation		X
<b>Future Land Use</b>				
Residents—Onsite	Groundwater	Ingestion Dermal Absorption Inhalation (Indoor Volatile)	X X X	
	Subsurface Soil	Ingestion Dermal Absorption	X X	

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exposure period; if exposures occur for a long period of time, a body weight of 70 kg (154 lb) is used. The averaging time depends on the type of health effect being assessed. For noncancer effects, the intake is calculated by averaging the intake over the period of exposure; for cancer, the intake is calculated by prorating the total cumulative dose over a lifetime.

The exposure (or intake) is normalized for time and body weight and is expressed as milligrams of chemical per kilogram of body weight per day (mg/kg per day). A general equation for intake is:

$$\text{Intake} = \frac{\text{Concentration} \times \text{Contact Rate} \times \text{Exposure Frequency} \times \text{Exposure Duration}}{\text{Body Weight} \times \text{Averaging Time}}$$

EPA guidance states that the quantification of exposure at a site should be based on an estimate of the "reasonable maximum exposure" expected under both current and future land use conditions (EPA, 1989b). The reasonable maximum exposure is defined as the "highest exposure that is reasonably expected to occur at a site" (EPA, 1989b). The intent of the reasonable maximum exposure is to estimate a conservative exposure case (i.e., well above the average case) that is still within the range of possibilities. Recent EPA guidance recommends that an estimate of average exposure also be identified. This is accomplished by applying average exposure assumptions along with average contaminant concentrations.

For this assessment, the health effects of the chemicals of potential concern have been separated into those that have cancer effects and those that have noncancer effects. The intake of a chemical evaluated for cancer health effects (i.e., lifetime average chemical intake) is calculated by prorating the total cumulative dose of the chemical over an averaging time of an entire life span, assumed to be 70 years. By convention, the assumption is for a "typical" individual with a lifetime average body weight of 70 kg (i.e., an adult male). The selection of an averaging time that spans a lifetime is based on EPA guidance, which states that the approach for cancer-causing chemicals is based on the assumption that a high dose received over a short period of time is equivalent to a corresponding low dose spread over a lifetime (EPA, 1989b). For noncancer effects, intake is prorated over the duration of exposure.

### *Current Use*

The site is a mixture of commercial and residential areas as discussed in an earlier chapter. The onsite area consists of three facilities, namely Granite Knitting Mills, Gentle Cleaners, and Parkside Apartments.



Table 6-7 summarizes the exposure pathways that will be addressed in this risk assessment. In reviewing the surface soil data (0 to 2 feet) for Granite Knitting Mills, Gentle Cleaners, and Parkside Apartments, it was noted that the Granite Knitting Mills facility has asphalt or buildings covering all available surface soil. This implies that there is no potential for direct contact with surface soil at the facility. In addition, the field GC and CLP surface soil data at the other two facilities show all nondetects except for one sample with tetrachloroethene (PCE) at 23  $\mu\text{g}/\text{kg}$ . For comparison, the value of 23  $\mu\text{g}/\text{kg}$  of PCE is about 1,000 times lower than the calculated level of concern for residential ingestion of PCE in soil (Roy Smith, "Risk-Based Concentration Table," Third Quarter 1992). In light of these results, the potential risks associated with current direct contact with surface soil are discussed in a qualitative rather than quantitative manner.

The air pathways from either outdoor exposure to VOCs in soil or from VOCs in soil entering the home also are being treated in a qualitative manner. This is based on the relatively low VOC concentration in surface soil and the statement in RAGs Part B (EPA, 1991) that "for many undisturbed sites with vegetative cover such as those found in areas of residential land use, air pathways are relatively minor contributors of risk."

Groundwater is not currently used onsite but there are a number of residential wells offsite. Residential use of contaminated groundwater for drinking water leads to exposure through direct ingestion of water and drinks made from water; inhalation of vapors from showering, bathing, washing, and other indoor uses; and dermal absorption from showering and bathing. Only reasonable maximum (or upper bound) exposure assumptions will be used for the residential use of groundwater scenario because of the limited range for the parameters needed to calculate intake.

**Ingestion Exposure.** Table 6-8 summarizes the intake equation for ingesting groundwater. The upper bound of intake-factor values for the ingestion rate, exposure frequency, exposure duration, body weight, and averaging time are used to calculate reasonable maximum exposure. The actual amount ingested may vary and is a result of several factors, including age, activity, dietary patterns, and climate. Ingestion results from ingestion of tap water; inadvertent ingestion during showering; and ingestion of fluids containing tap water (e.g., tea) and food made with tap water (e.g., soup). Examples of individuals who might consume more water than typical adults include formula-fed infants, adults engaged in strenuous physical activities, and individuals who are ill.

The National Academy of Science (NAS) calculated the average water consumption per capita per day as 1.63 liters but adopted 2 liters for use as the amount of water consumed (NAS 1977) per day. The EPA uses standard water-intake rates of 2 liters/day for adults and 1 liter/day for infants (EPA, 1989b). Although individuals may ingest more than this amount, this baseline risk assessment uses the EPA assumption of 2 liters/day for a 70-kg adult for calculating reasonable maximum exposure through ingestion.

**Table 6-8**  
**EXPOSURE ASSUMPTIONS FOR INGESTION OF GROUNDWATER**  
**RESIDENTIAL SCENARIO**

Assumption	Upper Bound	References
Ingestion Rate (l/day) Child Adult	1 2	EPA 1989a.
Exposure Frequency (day/year) Child and Adult	350	EPA 1991a.
Exposure Duration (years) Child Adult	6 24	EPA 1991a.
Body Weight (kg) Child Adult	15 70	EPA 1991a.
Averaging Time (years) Noncancer Child Adult Cancer Child and Adult	6 24 70	EPA 1991a.

Equation for intake of chemicals in drinking water (from EPA 1989b):

$$\text{Intake (mg/kg-day)} = \frac{\text{CW} \times \text{IR} \times \text{EF} \times \text{ED}}{\text{BW} \times \text{AT}}$$

Where:

- CW = Chemical Concentration in Water (mg/liter)
- IR = Ingestion Rate (liters/day)
- EF = Exposure Frequency (days/year)
- ED = Exposure Duration (years)
- BW = Body Weight (kg)
- AT = Averaging Time (period over which exposure is averaged-days)

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**Inhalation Exposure.** Individuals can be exposed to VOCs released from tap water into the air from showers, baths, toilets, dishwashers, and washing machines, as well as during cooking. There is no standard intake assumption for estimating residential exposure to VOCs released from tap water. Several variables can affect the degree of exposure. For example, exposure that takes place in the shower is affected by the temperature of the water, the type of shower head, the duration of the shower, the size of the shower space, and the air-exchange rate between the shower and the rest of the bathroom or house. In addition, individual chemicals volatilize at different rates.

Recent studies suggest that exposure to volatile chemicals from inhalation can be as great as, or greater than, exposure from ingestion alone. McKone (1987) estimated inhalation exposure to volatile compounds in a household over a 24-hour period, using a 3-compartment model to simulate the release of seven VOCs into a home. He developed indoor-air profiles to reflect changing concentrations over time and combined the data with estimates of individual activities to estimate exposure. McKone estimated that inhalation exposure would be 1.5 times (average) to 6 times (upper-bound estimate) that of ingestion.

By analyzing exhaled breath, Jo et al. (1990a) conducted studies to determine the inhalation and dermal uptake of chloroform from showering with chlorinated tap water. Subjects showered normally, and their exhaled breath represented both dermal and inhalation intake. In a companion experiment, the subjects wore rubber clothes and boots to preclude dermal contact but to allow inhalation exposure. By comparing the two showering experiments, Jo et al. concluded that the intake of chloroform during showering was approximately equal for inhalation and dermal routes.

Table 6-9 lists the factors needed to calculate the intake from inhaling vapor-phase chemicals during indoor use of contaminated groundwater during showering. Essential to the successful estimation of inhalation intake is the concentration of volatiles in indoor air. Because there are no chemical-specific or site-specific measurements of indoor volatiles for this site, the Foster and Chrostowski Shower Model (1987) is used here. It gives the relationship between the concentration of a volatile in water and in the bathroom air during showering. A copy of this showering model paper and input parameter modifications is presented in Appendix G.

**Dermal Exposure.** Another potential route of exposure associated with residential use of contaminated drinking water is dermal absorption of contaminants. Dermal absorption can occur during bathing, showering, food preparation, and dishwashing.

Skin is a relatively good barrier separating humans from chemicals in their environment. Absorption of a chemical requires passage through the outer skin layer, or the stratum corneum, which is the rate-limiting (i.e., slowest) step in dermal absorption. Once chemicals pass through the stratum corneum, they move into the blood stream with relative ease. Many factors can influence the absorption of

**Table 6-9**  
**EXPOSURE ASSUMPTIONS FOR INHALATION OF**  
**VAPOR PHASE CHEMICALS DURING SHOWERING**  
**RESIDENTIAL SCENARIO**

Assumption	Upper Bound	References
Inhalation Rate (m <sup>3</sup> /hr) Child and Adult	0.83	Foster and Chrostowski, 1987. The inhalation rate of the child is assumed to be equal to the adult during this short time period as a conservative estimate.
Exposure Time (hr/day) Child and Adult	0.2	EPA 1989a.
Exposure Frequency (days/year) Child and Adult	350	EPA 1991a.
Exposure Duration (years) Child Adult	6 24	EPA 1991a.
Body Weight (kg) Child Adult	15 70	EPA 1991a.
Averaging Time (years) Noncancer Child Adult Cancer Child and Adult	6 24 70	EPA 1991a.

Equation for inhalation of residential vapor-phase chemicals (from EPA 1989b):

$$\text{Intake (mg/kg-day)} = \frac{\text{CA} \times \text{IR} \times \text{ET} \times \text{EF} \times \text{ED}}{\text{BW} \times \text{AT}}$$

Where:

- CA = Chemical Concentration in Air (mg/m<sup>3</sup>)
- IR = Inhalation Rate (m<sup>3</sup>/hour)
- ET = Exposure Time (hours/day)
- EF = Exposure Frequency (days/year)
- ED = Exposure Duration (years)
- BW = Body Weight (kg)
- AT = Averaging Time (period over which exposure is averaged—days)

chemicals across the skin layer. They include the health of the skin, location of the area of exposed skin, hydration of the skin, length of exposure, molecular size of the chemical, lipid solubility, thickness of the skin, skin temperature, and type of solvent in which the solute (contaminant) is dissolved. Because dermal absorption is a complex activity affected by many factors, precisely predicting exposures from this route is difficult.

Table 6-10 presents the equation and the factors needed to calculate the absorbed dose (the term used for the intake of chemicals through the skin) for dermal contact with chemicals in household water. The value that is frequently unavailable or estimated in this equation is the permeability constant because this value has been measured for only a few compounds. The permeability constants listed in Table 6-11 are either measured or estimated using the approach described in *Guidance for Dermal Exposure Assessment* (EPA, 1992).

**Comparison of Three Routes of Exposure (For Volatile Contaminants).** Several researchers have compared the relative contribution of ingestion, inhalation, and dermal absorption of volatile contaminants with the total intake of a contaminant from residential use of drinking water.

Brown et al. (1984) calculated the relative contribution (percentage) of dermal and oral exposure to dose, using a 70-kg adult bathing 15 minutes, 80 percent immersed, with 1.8 m<sup>2</sup> skin surface area, and consuming 2 liters of water per day as follows:

Compound	Concentration (mg/l)	Percentage Relative Contribution	
		Dermal (%)	Oral (%)
Toluene	0.005	67	33
	0.10	63	37
	0.5	59	41

Shehata (1985) found the percentage of relative contribution for the inhalation, oral, and dermal routes unchanged for toluene in the concentration range of 0.005 to 50.0 mg/l for the case of a child's total body burden in summer (rural) as follows:

Compound	Chemical Concentration in Drinking Water (mg/l)	Percentage Relative Contribution		
		Inhalation (%)	Oral (%)	Dermal (%)
Toluene	0.005 to 50.0	22.0	45.0	32.0

**Table 6-10**  
**EXPOSURE ASSUMPTIONS FOR DERMAL CONTACT**  
**WITH CHEMICALS IN WATER DURING SHOWERING**  
**RESIDENTIAL SCENARIO**

Assumption	Upper Bound	References
Skin Surface Area (m <sup>2</sup> ) Child Adult	0.918 2.28	EPA 1989a.
Permeability Constant (cm/hr)	Chemical-specific	
Exposure Time (hr) Child and Adult	0.2	EPA 1989a.
Exposure Frequency (days/year) Child & Adult	350	EPA 1991a.
Exposure Duration (years) Child Adult	6 24	EPA 1991a.
Body Weight (kg) Child Adult	15 70	EPA 1991a.
Averaging Time (years) Noncancer Child Adult Cancer Child and Adult	  6 24  70	EPA 1991a.

Equation for dermal contact with chemicals in water during showering (from EPA 1989b):

$$\text{Absorbed Dose (mg/kg-day)} = \frac{\text{CW} \times \text{SA} \times \text{PC} \times \text{ET} \times \text{EF} \times \text{ED} \times \text{CF}}{\text{BW} \times \text{AT}}$$

Where:

- CW = Chemical Concentration in Water (mg/l)
- SA = Skin Surface Area Available for Contact (m<sup>2</sup> or cm<sup>2</sup>)
- PC = Chemical-Specific Dermal Permeability Constant (cm/hr)
- ET = Exposure Time (hours/day)
- EF = Exposure Frequency (days/year)
- ED = Exposure Duration (years)
- CF = Volumetric Conversion Factor for Water (1 liter/1,000 cm<sup>3</sup>)
- BW = Body Weight (kg)
- AT = Averaging Time (period over which exposure is averaged-days)

**Table 6-11  
AQUEOUS DERMAL PERMEABILITY CONSTANTS**

Contaminants Detected in Groundwater	Dermal Permeability Constant Kp (cm/hr) <sup>a</sup>	
	Experimentally Measured Values	Predicted Values
Benzene	0.11	
Bromodichloromethane		0.0058
Carbon Disulfide	0.50	
Chloroform	0.13	
1,1-Dichloroethane		0.0089
trans-1,2-Dichloroethene		0.010
Ethylbenzene	1.0	
Tetrachloroethene	0.37	
Toluene	1.0	
1,1,1-Trichloroethane		0.017
Trichloroethene	0.23	

<sup>a</sup>From U.S. EPA *Dermal Exposure Assessment: Principles and Applications*. Office of Health and Environmental Assessment. U.S. EPA/600/8-91/011B. January 1992. Table 5-8. Measured Kps were given priority over predicted values.

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Hall et al. (1989) compared the relative contribution of ingestion, inhalation, and dermal pathways to the percentage of the lifetime equivalent exposure to trans-1,2-dichloroethene attributable to each of the three pathways as follows:

Exposure	Percentage Relative Contribution		
	Ingestion	Inhalation	Dermal
Best Estimate	31	41	28
Upper Bound	21	65	14

Brown and Hattis (1989) calculated the relative contribution of different routes of exposure to the absorbed dose of ethylbenzene, tetrachloroethene, and trichloroethylene in drinking water. The oral-dose values assumed an intake of 2 liters/day, and the inhalation dose was based on the indoor air model of McKone (1987). Ethylbenzene had approximately the same intake from all three routes when conditions were set to allow for maximum dermal absorption. The parameters that were changed to reflect the minimum and maximum conditions of dermal uptake include the percentage of fat in skin and blood as follows:

Compound		Percentage of Total Dose		
		Oral	Inhalation	Dermal
Ethylbenzene	Min.	37.9	54.5	7.6
	Max.	26.0	37.3	36.7
Tetrachloroethylene	Min.	41.7	56.9	1.5
	Max.	34.3	46.8	18.9
Trichloroethylene	Min.	40.6	58.2	1.2
	Max.	34.3	49.2	16.5

Jo et al. (1990b) estimated the chloroform dose from showering to be 0.24  $\mu\text{g}/\text{kg}$  per day for inhalation and 0.22  $\mu\text{g}/\text{kg}$  per day for the dermal route, for a total of 0.46  $\mu\text{g}/\text{kg}$  per day. The chloroform-water concentration used to estimate ingestion intake was a value proportional to the mean shower-air concentration; for 2 liters/day of water ingestion, the chloroform intake was 0.7  $\mu\text{g}/\text{kg}$  per day, or roughly the same daily intake as the combined dose from inhalation and dermal exposure.



Another potential exposure pathway under current land use is the outdoor inhalation of VOCs from contaminants in surface soil and the indoor inhalation of VOCs entering the home. The low levels of VOCs in surface soil imply that this is not a viable source for either ingestion or inhalation of VOCs outdoors. The potential might exist for subsurface VOCs to migrate into homes along fractures in the soil and through cracks in home foundations. Again, the relatively low levels of VOCs in subsurface soil indicate that this is not a potential pathway of great concern.

### *Future Use*

Table 6-7 shows that for future land use, residents could be potentially exposed to groundwater onsite and subsurface soils following excavation during construction. The intake equations and assumptions for residential use of groundwater are the same as for current offsite groundwater use. To address potential exposure to contaminants in subsurface soil, it is assumed that the soils from 2 to 10 feet could be brought to the surface during construction with subsequent direct contact by residents. This direct contact is treated in a conservative fashion (i.e., likely to produce higher risks) by assuming that the child and adult could be in contact for 6 and 24 years, respectively, and that the concentration of VOCs will remain constant throughout this period. The assumption of a constant concentration is highly unlikely because of the volatility of the contaminants and other degradation effects.

The exposure assumptions and intake equations for ingestion of soil and dermal contact are given in Tables 6-12 and 6-13. The exposure frequency of contact with soil is estimated on the basis of the probable weather conditions at the site inhibiting outside activity during the cold months.

Separate intake values are calculated for a child and adult as in the residential groundwater use scenario. However, new guidance in Part B of RAGs (Human Health Evaluation Manual, Part B: "Development of Risk-Based Preliminary Remediation Goals") gives an alternative method for calculating the risk from ingestion of soil:

Because the soil ingestion rate is different for children and adults, the risk due to direct ingestion of soil is calculated using an age-adjusted ingestion factor. The age-adjusted soil ingestion factor ( $IF_{\text{soil/adj}}$ ) takes into account the difference in daily soil ingestion rates, body weights, and exposure durations for two exposure groups—children of one to six years and others of seven to 31 years. Exposure frequency (EF) is assumed to be identical for the two exposure groups. For convenience, this factor is calculated separately as a time-weighted soil intake, normalized to body weight, that can then be substituted in the total intake equation. Calculated in this manner, the factor leads to more protective risk-based concentration compared to an adult-only assumption.

**Table 6-12  
EXPOSURE ASSUMPTIONS FOR INCIDENTAL INGESTION OF CONTAMINANTS IN SOIL  
RESIDENTIAL SCENARIO**

Assumption	Average	Reasonable Maximum	References and Notes
Ingestion Rate (g/day) Toddler (age 1-6 years) Adult	0.2 0.1	0.2 0.1	EPA 1989a.
Fraction Ingested from Contaminated Source (unitless)	1	1	Assumed to be ingested from the site only.
Exposure Frequency (days/yr) Toddler (age 1-6 years) Adult	175 175	350 350	Values for reasonable maximum are for standard residential scenario as per request of EPA Region III.
Exposure Duration (years) Toddler (age 1-6 years) Adult	6 9	6 24	EPA 1991a. Nine years is the 50th percentile residence time in a particular home, 24 years the upper 90th.
Body Weight (kg) Toddler (age 1-6 years) Adult	15 70	15 70	EPA 1991a.
Averaging Time (years) Carcinogens  Noncarcinogens (age 1-6 years) Adult	70  6 9	70  6 24	EPA 1991a. Seventy-year lifetime for carcinogenic effects.  Period of exposure is the exposure duration for noncarcinogenic effects.

Equation for Ingestion of Chemicals in Soil (from EPA 1989b):

$$\text{Intake (mg/kg-day)} = \frac{\text{CS} \times \text{IR} \times \text{CF} \times \text{FI} \times \text{EF} \times \text{ED}}{\text{BW} \times \text{AT}}$$

Where:

- CS = Chemical Concentration in Soil (mg/kg)
- IR = Ingestion Rate (g/day)
- CF = Conversion Factor ( $10^{-6}$  kg/mg)
- FI = Fraction Ingested from Contaminated Source (unitless)
- EF = Exposure Frequency (days/year)
- ED = Exposure Duration (years)
- BW = Body Weight (kg)
- AT = Averaging Time (period over which exposure is averaged—days)

**Table 6-13  
EXPOSURE ASSUMPTIONS FOR DERMAL CONTACT WITH CHEMICALS IN SOIL  
RESIDENTIAL SCENARIO**

Assumption	Average	Reasonable Maximum	References and Notes
Skin Surface Area (m <sup>2</sup> /event) Toddler (age 1-6 years) Adult	0.24 0.31	0.47 0.51	EPA 1989a. Average exposure includes surface area of hands, lower arms, and head. Upper-bound exposure includes surface area of hands, lower arms, head, lower legs, and feet (toddler only).
Soil to Skin Adherence Factor (mg/cm <sup>2</sup> ) Commercial Potting Soil—hands	0.2	1.0	EPA 1992a.
Absorption Factor (unitless)	Chemical-Specific	Chemical-Specific	See text.
Exposure Frequency (days/year) Toddler (age 1-6 years) Adult	175 175	350 350	Values for reasonable maximum are for standard residential scenario as per request of EPA Region III.
Exposure Duration (years) Toddler (age 1-6 years) Adult	6 9	6 24	EPA 1991a. Nine years is the 50th percentile residence time in a particular home, 24 years the upper 90th.
Body Weight (kg) Toddler (age 1-6 years) Adult	15 70	15 70	EPA 1991a.
Averaging Time (years) Carcinogens  Noncarcinogens Toddler (age 1-6 years) Adult	70  6 9	70  6 24	EPA 1991a. Seventy-year lifetime for carcinogenic effects.  Period of exposure is the exposure duration for noncarcinogenic effects.

Equation for Dermal Contact With Chemicals in Soil (EPA 1989b):

$$\text{Absorbed Dose (mg/kg-day)} = \frac{\text{CS} \times \text{CF} \times \text{SA} \times \text{AF} \times \text{ABS} \times \text{EF} \times \text{ED}}{\text{BW} \times \text{AT}}$$

Where:

- CS = Chemical Concentration in Soil/Sediment (mg/kg)
- CF = Conversion Factor (10<sup>-6</sup> kg/mg)
- SA = Skin Surface Area Available for Contact (m<sup>2</sup> or cm<sup>2</sup>/event)
- AF = Soil to Skin Adherence Factor (mg/cm<sup>2</sup>)
- ABS = Absorption Factor (unitless)
- EF = Exposure Frequency (days/year)
- ED = Exposure Duration (years)
- BW = Body Weight (kg)
- AT = Averaging Time (period over which exposure is averaged—days)

This new method combines the two age groups into one continuous receptor for the entire duration of living in a residence (30 years). Calculating separate intakes for the child and adult does allow for totaling the cumulative risk for each age group and is the approach that will be followed in this risk assessment.

The chemical-specific dermal absorption factors for the chemicals of concern are approximated to be 10 to 25 percent since no literature values could be found. Using 25 percent as the dermal absorption factor for VOCs in soil may lead to upper bound (i.e., conservative) estimates of risk.

## 6.8 Risk Characterization

The potential for adverse human health effects posed by the contaminants of potential concern is a function of both the toxicity of the contaminants and the possibility of exposure. This section combines the dose-response characteristics with the exposure assessment to characterize the risk of carcinogenic and noncarcinogenic effects in the potentially exposed population. The methodologies used to assess carcinogenic and noncarcinogenic risks are discussed first, followed by quantitative calculations associated with current land use and potential future use.

### 6.8.1 Assessment Methodology for Carcinogenic Risk

For carcinogens, risks are estimated as the incremental increase in the probability of an individual developing cancer over a lifetime as a result of exposure to a potential carcinogen. This incremental increase in probability is typically expressed as excess individual lifetime cancer risk. This section describes the methodology for estimating cancer risks from exposure to either a single carcinogen or multiple carcinogens.

#### *Cancer Risks From Exposure to a Single Carcinogen*

The "one-hit" equation below can be used to describe excess lifetime cancer risk from exposure to one carcinogen (EPA, 1989b):

$$\text{Risk} = 1 - \exp^{-(\text{SF} \times \text{CDI})}$$

Where:

- Risk = Excess lifetime cancer risk as a unitless probability
- SF = Cancer slope factor (kg-day/mg)
- CDI = Chronic daily intake averaged over a lifetime (mg/kg-day)

Where the excess lifetime cancer risk is less than 0.01, it can generally be assumed that the dose-response relationship will be in the linear low-dose portion of the dose-response curve. In such cases, the slope factor is a constant, and excess cancer risk is calculated by multiplying the slope factor directly by the chronic daily intake (EPA, 1989b):

$$\text{Risk} = \text{SF} \times \text{CDI}$$

### ***Cancer Risks from Exposure to Multiple Carcinogens***

Many exposure scenarios involve potential exposure to more than one carcinogen. In assessing the potential for carcinogenic effects from exposure to multiple carcinogens, the assumption in the absence of information on synergistic or antagonistic effects is that carcinogenic risks are additive. This approach is based on the EPA's *Guidelines for Health Risk Assessment of Chemical Mixtures* (EPA, 1986b) and *Guidelines for Carcinogen Risk Assessment* (EPA, 1986a).

### ***Assessment Methodology for Noncarcinogenic Risk***

The potential for noncarcinogenic effects on health resulting from exposure to a contaminant is evaluated by comparing an intake level (mg/kg-day) over a specified time period with an RfD (mg/kg-day) derived for a similar exposure period. This ratio of exposure to toxicity is called a hazard quotient (HQ) and is a unitless number. The HQ is a numerical indicator of the transition between acceptable and unacceptable exposure levels, and when the HQ exceeds 1, the result may be adverse health effects. Unlike the excess lifetime cancer risk, this value is not a probability. It does not predict the incidence of adverse noncarcinogenic effects.

When the potential exists for exposure to more than one chemical, the potential for noncarcinogenic effects posed by multiple chemicals can be screened by using a hazard index approach. This approach, which is based on the EPA's *Guidelines for Health Risk Assessment of Chemical Mixtures* (EPA, 1986b), assumes dose additivity and sums the ratios of the daily intakes of individual chemicals to their reference doses. This sum is called the hazard index.

As for the HQ, when the hazard index exceeds 1, it indicates a greater possibility that adverse health effects might occur. Any single chemical with an estimated daily intake greater than the corresponding RfD will cause the hazard index to exceed 1.

For multiple chemical exposures, the hazard index can exceed 1 even if no single chemical exposure exceeds the RfD for that chemical. The assumption of additivity is applied most appropriately to chemicals that induce the same effect by the same mechanism in the same target organ. If the hazard index exceeds 1, the chemicals in the mixture may be segregated by critical effect or target organ, and separate indexes may be derived for each effect or target organ. If any of these separate indexes exceed 1, then there may be a concern about potential health effects.

## 6.8.2 Risk Characterization Results

This section presents the results of the risk characterization for subsurface soil and groundwater exposure at the site. The noncarcinogenic risk and excess lifetime cancer risk for each exposure scenario associated with current and potential future land use are presented in summary tables. Complete risk characterization tables for all chemicals of potential concern are presented in Appendix G.

The EPA recently released a guidance memorandum on risk characterization titled *Guidance on Risk Characterization for Risk Managers and Risk Assessors* (EPA, 1992). This new guidance describes how to present risk assessment results in EPA reports in order to show a full and complete analysis of risk and to promote greater consistency and comparability in risk assessments.

One of the recommendations of the risk characterization guidance is to include an estimate of central tendency (or average) exposure. An estimate of average risk for each scenario has been included by coupling average exposure assumptions with average contaminant concentration (arithmetic mean). The uncertainties associated with both the average risk and reasonable maximum risk will be presented in the next section.

## 6.8.3 Residential Groundwater Use

Table 6-14 presents the summary of risk characterization results for current offsite and potential future onsite use of groundwater. These are summary results of individual calculations of each scenario using the intake equations and assumptions discussed in the exposure assessment section. The complete calculations are in Appendix E.

The current offsite residential well use includes the well locations shown in Figure 4-7 and discussed in Section 2.5. The summary statistics are given in Table 6-3. As can be seen from the maximum concentrations detected, none of the VOCs are above their MCLs, with most being near their detection limit.

The excess lifetime cancer risk and noncarcinogenic risk from the residential use of this groundwater is shown in Table 6-14. The overall observation is that the low concentrations of the VOCs result in low risk levels. The excess lifetime cancer risk for the child calculated using reasonable maximum exposure assumptions and upper-bound concentrations is  $1 \times 10^{-6}$ , which is at the "point of departure" or target risk level established by EPA. This risk level is 100 times lower than the cumulative excess lifetime cancer risk level of  $10^{-4}$  established as a point of departure in OSWER Directive 9355.0-30 "Role of the Baseline Risk Assessment in Superfund Remedy Selection Decisions," April 22, 1991. "Where the cumulative carcinogenic site risk to an individual based on reasonable maximum exposure for both current and future land use is less than  $10^{-4}$ , and the non-carcinogenic hazard quotient is less than 1, action generally is not warranted unless there are adverse environmental impacts." The excess lifetime cancer risk for the adult is slightly higher than for the child at  $2 \times 10^{-6}$ .

Table 6-14

**SUMMARY OF RISK CHARACTERIZATION RESULTS  
RESIDENTIAL GROUNDWATER USE**

Scenario/ Location	Exposure Pathway	Hazard Index		Excess Lifetime Cancer Risk	
		Average	Reasonable Maximum	Average	Reasonable Maximum
<b>Current/Offsite</b>					
Child	Ingestion	0.01	0.01	$4 \times 10^{-7}$	$6 \times 10^{-7}$
	Dermal	0.01	0.01	$2 \times 10^{-7}$	$3 \times 10^{-7}$
	Inhalation	0.03	0.04	$5 \times 10^{-7}$	$5 \times 10^{-7}$
	Total	0.05	0.06	$1 \times 10^{-6}$	$1 \times 10^{-6}$
Adult	Ingestion	0.01	0.01	$8 \times 10^{-7}$	$1 \times 10^{-6}$
	Dermal	0.003	0.004	$5 \times 10^{-7}$	$6 \times 10^{-7}$
	Inhalation	0.007	0.009	$4 \times 10^{-7}$	$5 \times 10^{-7}$
	Total	0.02	0.02	$2 \times 10^{-6}$	$2 \times 10^{-6}$
<b>Future/Onsite</b>					
Child	Ingestion	0.03	0.05	$8 \times 10^{-7}$	$2 \times 10^{-6}$
	Dermal	0.01	0.03	$5 \times 10^{-7}$	$1 \times 10^{-6}$
	Inhalation	0.09	0.2	$4 \times 10^{-7}$	$5 \times 10^{-7}$
	Total	0.1	0.3	$2 \times 10^{-6}$	$4 \times 10^{-6}$

**Table 6-14**  
**SUMMARY OF RISK CHARACTERIZATION RESULTS**  
**RESIDENTIAL GROUNDWATER USE**

Scenario/ Location	Exposure Pathway	Hazard Index		Excess Lifetime Cancer Risk	
		Average	Reasonable Maximum	Average	Reasonable Maximum
Adult	Ingestion	0.01	0.02	$1 \times 10^{-6}$	$3 \times 10^{-6}$
	Dermal	0.01	0.01	$1 \times 10^{-6}$	$2 \times 10^{-6}$
	Inhalation	0.02	0.03	$4 \times 10^{-7}$	$4 \times 10^{-7}$
	Total	0.04	0.06	$2 \times 10^{-6}$	$5 \times 10^{-6}$
<b>Future/Onsite—Shallow Aquifer Scenario (Highest Concentration of PCE)</b>					
Child	Ingestion		2.1		$9 \times 10^{-5}$
	Dermal		1.4		$6 \times 10^{-5}$
	Inhalation		4.9		$8 \times 10^{-6}$
	Total		8		$2 \times 10^{-4}$
Adult	Ingestion		0.9		$2 \times 10^{-4}$
	Dermal		0.8		$1 \times 10^{-4}$
	Inhalation		1.0		$7 \times 10^{-6}$
	Total		3		$3 \times 10^{-4}$

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The noncarcinogenic risk for current offsite residential groundwater use is well below the "point of departure" or target risk level of hazard index equal to 1 as established by EPA. As in the earlier discussion of the relative contribution of different exposure routes, the contribution of dermal and inhalation risk are about the same as from ingestion. The hazard index of 0.06 for the child is larger than the noncarcinogenic risk to adults. The average risk is only slightly lower than the reasonable maximum risk due to the limited range of exposure assumptions and concentrations.

Table 6-14 also presents the risks associated with the potential future onsite residential use of groundwater. The excess lifetime cancer risks and noncarcinogenic risks are roughly three times the levels seen for current offsite use of groundwater. The excess lifetime cancer risk for the child is  $4 \times 10^{-6}$  while for the adult, it is  $5 \times 10^{-6}$  with both levels still about at the "point of departure" or target risk level. The noncarcinogenic risk of hazard index equal to 0.3 for the child is 3 times lower than EPA's target risk level of hazard index equal to 1. The adult hazard index level of 0.06 is 5 times lower than the child.

Potential future onsite residential use of groundwater from the shallow aquifer where water infiltration was collected during a packer test was calculated. This represents an extreme case since the flow rate is low, and this is a grab sample instead of a sample from a properly flushed monitoring well. Only 1 VOC was detected—PCE at a concentration of  $330 \mu\text{g/l}$ .

Table 6-14 also presents the risks associated with the potential future onsite residential use of groundwater from the shallow aquifer. It can be seen that the excess lifetime cancer risks of  $2 \times 10^{-4}$  for the child and  $3 \times 10^{-4}$  for the adult are at the upper end of the target risk level established by EPA. The noncarcinogenic hazard quotient is greater than the target risk level of 1 at 8 for the child and 3 for the adult.

#### **6.8.4 Direct Contact with Subsurface Soil**

Table 6-15 gives the summary of risk characterization results for potential future contact by onsite residents with subsurface soil following excavation. The assumptions used to predict risk from this scenario are very conservative. For example, the concentration of VOCs in the soil are projected to be the same for extended periods of time (24 years for the adult), which is highly unlikely because of the volatility of the compounds. Another assumption is that all of the soil that the resident would come into contact with would be from the excavated subsurface soil. In addition, the dermal absorption factor of 25 percent may be much higher than the actual value, and the use of 350 days/year of contact (as per recommendation by Region III) does not account for snow days or frozen ground.

Table 6-15

**SUMMARY OF RISK CHARACTERIZATION RESULTS  
POTENTIAL FUTURE CONTACT WITH SUBSURFACE SOIL**

Location	Exposure Pathway	Hazard Index		Excess Lifetime Cancer Risk	
		Average	Reasonable Maximum	Average	Reasonable Maximum
<b>Parkside Apartments</b>					
Child	Ingestion	<0.001	<0.001	$2 \times 10^{-9}$	$6 \times 10^{-9}$
	Dermal	<0.001	0.001	$4 \times 10^{-10}$	$4 \times 10^{-8}$
	Total	<0.01	<0.01	$2 \times 10^{-9}$	$5 \times 10^{-8}$
Adult	Ingestion	<0.001	<0.001	$3 \times 10^{-10}$	$3 \times 10^{-9}$
	Dermal	<0.001	<0.001	$2 \times 10^{-10}$	$3 \times 10^{-8}$
	Total	<0.01	<0.01	$5 \times 10^{-10}$	$3 \times 10^{-8}$
<b>Granite Knitting Mills</b>					
Child	Ingestion	0.001	0.009	$5 \times 10^{-8}$	$4 \times 10^{-7}$
	Dermal	<0.001	0.05	$1 \times 10^{-8}$	$2 \times 10^{-6}$
	Total	<0.01	0.06	$6 \times 10^{-8}$	$2 \times 10^{-6}$
Adult	Ingestion	<0.001	<0.001	$8 \times 10^{-9}$	$2 \times 10^{-7}$
	Dermal	<0.001	0.01	$5 \times 10^{-9}$	$2 \times 10^{-6}$
	Total	<0.01	0.01	$1 \times 10^{-8}$	$2 \times 10^{-6}$

Location	Exposure Pathway	Hazard Index		Excess Lifetime Cancer Risk	
		Average	Reasonable Maximum	Average	Reasonable Maximum
<b>Gentle Cleaners</b>					
Child	Ingestion	0.06	0.4	$3 \times 10^{-6}$	$2 \times 10^{-5}$
	Dermal	0.02	2.3	$7 \times 10^{-7}$	$1 \times 10^{-4}$
	Total	0.08	3	$4 \times 10^{-6}$	$1 \times 10^{-4}$
Adult	Ingestion	0.007	0.04	$5 \times 10^{-7}$	$7 \times 10^{-6}$
	Dermal	0.004	0.5	$3 \times 10^{-7}$	$9 \times 10^{-5}$
	Total	0.01	0.5	$1 \times 10^{-6}$	$1 \times 10^{-4}$

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The onsite residential locations are divided into three facilities, namely Parkside Apartments, Granite Knitting Mills, and Gentle Cleaners. The excess lifetime cancer risk for both the child and adult at Parkside Apartments are well below EPA's "point of departure" as shown in Table 6-15. The noncarcinogenic risk for both the child and adult at Parkside Apartments also is several orders of magnitude lower than EPA's target hazard index of 1.

The excess lifetime cancer risk from ingestion and dermal contact with subsurface soil at Granite Knitting Mills is at or near EPA's point of departure for the child and adult. The risk from this pathway should be added to the risk from potential future use of onsite groundwater to obtain the total risk from all pathways. The total excess lifetime cancer risk is still well below the cumulative risk level of  $10^{-4}$  referenced earlier. The noncarcinogenic risk for direct contact with subsurface soil at Granite Knitting Mills is low.

The excess lifetime cancer risk and noncarcinogenic risk for Gentle Cleaners are at least 10 times higher than either of the other two facilities. Tetrachloroethene is the primary contaminant leading to an excess lifetime cancer risk of  $1 \times 10^{-4}$  for the child and the same for the adult. The noncarcinogenic risk for the child at 3 is greater than the target hazard index of 1. The average risk calculated for both noncarcinogenic risk and excess lifetime cancer risk are about an order of magnitude lower than the calculated reasonable maximum risk levels.

## 6.9 Uncertainty Analysis

The procedures and inputs used to assess risks are subject to a wide variety of uncertainties. In general, the main sources of uncertainty include environmental chemistry, sampling and analysis, environmental parameter measurement, fate and transport modeling (not used for this assessment), exposure parameter estimation, and toxicological data.

Uncertainty in environmental sampling arises in part from the potentially uneven distribution of chemicals in the media sampled. This results in significant uncertainty as to the actual levels present. Environmental chemistry analysis errors can stem from several sources including the errors inherent in the analytical methods and characteristics of the media being sampled.

Uncertainties in toxicological data occur in extrapolating both from animals to humans, route to route, and high to low doses of exposure, as well as from the difficulties in assessing the toxicity of a mixture of chemicals. Chronic RfDs, which are usually appropriate for exposures greater than 7 years, have been used to calculate the noncarcinogenic risk for the child (1 to 6 years old). Oral toxicity values have been used to calculate dermal risk and for inhalation risk when inhalation toxicity values were unavailable. Table 6-6 presents a summary of sources of uncertainty in the toxicological data.

Uncertainties in the exposure assessment are related to estimates of how often an individual comes in contact with the chemical of potential concern, the period of time over which such exposure would occur, and the models used to estimate the concentrations of the chemicals of potential concern at the point of exposure.

The time of direct contact with onsite subsurface soil is probably overestimated, particularly because the contaminants are volatiles and will tend to dissipate when brought to the surface through construction work. The temperature and snow cover during the winter months also will inhibit direct contact.

The concentrations of VOCs used to characterize potential future onsite residential groundwater use are lower than what has been detected in the past by 3 to 10 times. In addition, the contaminants in soil could continue to leach to groundwater, which could elevate the VOC levels in groundwater. This would have a direct effect on the amount of calculated intake by either ingestion, dermal absorption, or inhalation and, thus, the excess lifetime cancer risk or noncarcinogenic risk. Because there is assumed to be a linear relationship between concentration and intake and between intake and risk, raising the concentration of VOCs in groundwater by 3 to 10 times would raise the risk associated with each VOC by the same amount. In addition, only VOCs entering the house from the showering scenario were considered; this ignores the contribution from other activities such as dishwashing, which could lead to underestimating the risk from this pathway.

It should be noted that the concentrations of PCE detected in the samples obtained from the full well at the GKM well and NPWA well S-9 during the RI were somewhat lower than those detected in the individual intervals sampled using the straddle packers. For the GKM well, the full-well sample had 3  $\mu\text{g/l}$  while the intervals had concentrations ranging from 16 to 330  $\mu\text{g/l}$ ; well S-9 had a full-well concentration of 5  $\mu\text{g/l}$  while the intervals contained from 8 to 17  $\mu\text{g/l}$ . The risk characterization results for the shallow aquifer packer tests with PCE at 330  $\mu\text{g/l}$  as given in Table 6-14 probably represent an extreme case.

Table 4-5 and Figure 4-13 shown PCE concentrations in the GKM well declining with continued pumping from 32  $\mu\text{g/l}$  to about 5  $\mu\text{g/l}$ ; this last concentration is similar to that of the full-well concentration. This decline suggests that the sample from the full-well is probably representative of the PCE concentration in the aquifer in the vicinity of this well when the aquifer has been flushed by continued pumping.

Because well S-9 was not pumped for as long as the GKM well before sampling, it is unknown if this same gradual decrease in PCE concentration may have occurred. If it did, then the full-well concentration of 5  $\mu\text{g/l}$  is representative of average groundwater conditions around the well. If not, it is possible that the sample was obtained from an interval of fractures containing relatively clean water and that the average concentration of PCE around well S-9 may actually be as high as 17  $\mu\text{g/l}$ . This possibility adds to the uncertainty associated with the risk calculations made using the analytical data obtained during the RI.

The summary tables of risk characterization results show that the average risks are on the order of 3 to 10 times lower than the reasonable maximum risks. This range is an indication of the uncertainty inherent in the exposure assumptions and contaminant concentrations and should be taken into account when decisions are to be made on the basis of the risk characterization results.

Monte Carlo calculations are provided when the excess lifetime cancer risk is greater than  $1 \times 10^{-4}$ . The Monte Carlo calculations are an extension of the process already displayed in this report of calculating risk using average concentrations coupled with average exposure assumptions and reasonable maximum concentrations with reasonable maximum exposure assumptions. The Monte Carlo calculations use probability distributions of the contaminant concentration and exposure assumptions in a random simulation to calculate risk hundreds or thousands of times.

The risk characterization results for direct contact with soil and for current residential use of groundwater are all  $1 \times 10^{-4}$  or less. The one scenario where the excess lifetime cancer risk is greater than  $1 \times 10^{-4}$  is for future use of groundwater from the shallow aquifer with the reasonable maximum risk calculated to be  $2 \times 10^{-4}$  for a child and  $3 \times 10^{-4}$  for an adult. This future use scenario is based on a single data point for the concentration term and, thus, is not well suited for a Monte Carlo simulation since the input parameters should be entered as a range of possible values along with the probability of occurrence, rather than a single value.

For comparative purposes, however, this future-use scenario was modeled using Monte Carlo simulation following the recommendations in the June 26, 1992, memo from Ray Smith titled "Monte Carlo risk assessment: a regional science policy issue." The complete paper and input parameter distributions are given in Appendix G. The excess lifetime cancer risk simulations using both Latin Hypercube and Monte Carlo modes for @RISK are also displayed in Figures 6-2 and 6-3. It is clear that the RME value of  $3 \times 10^{-4}$  for the adult is at the high end of the probability distribution and represents a "worst case" scenario.

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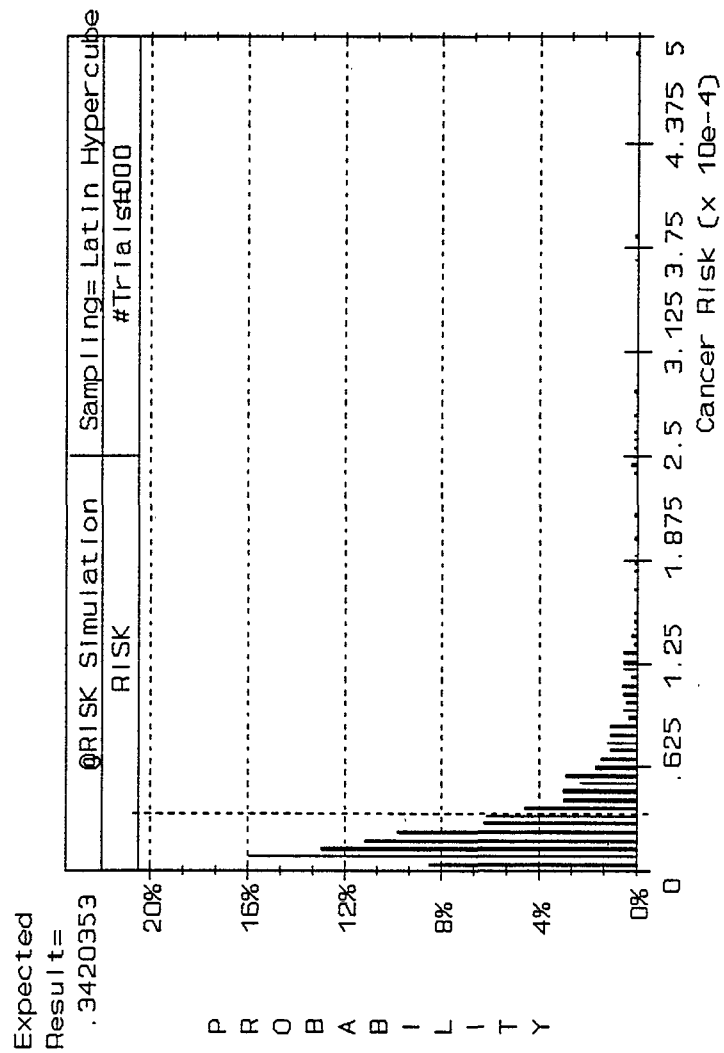


Figure 6-2  
 Excess Lifetime Cancer Risk  
 Simulation--Latin Hypercube  
 North Penn Area 1  
 Phase II RI/FS

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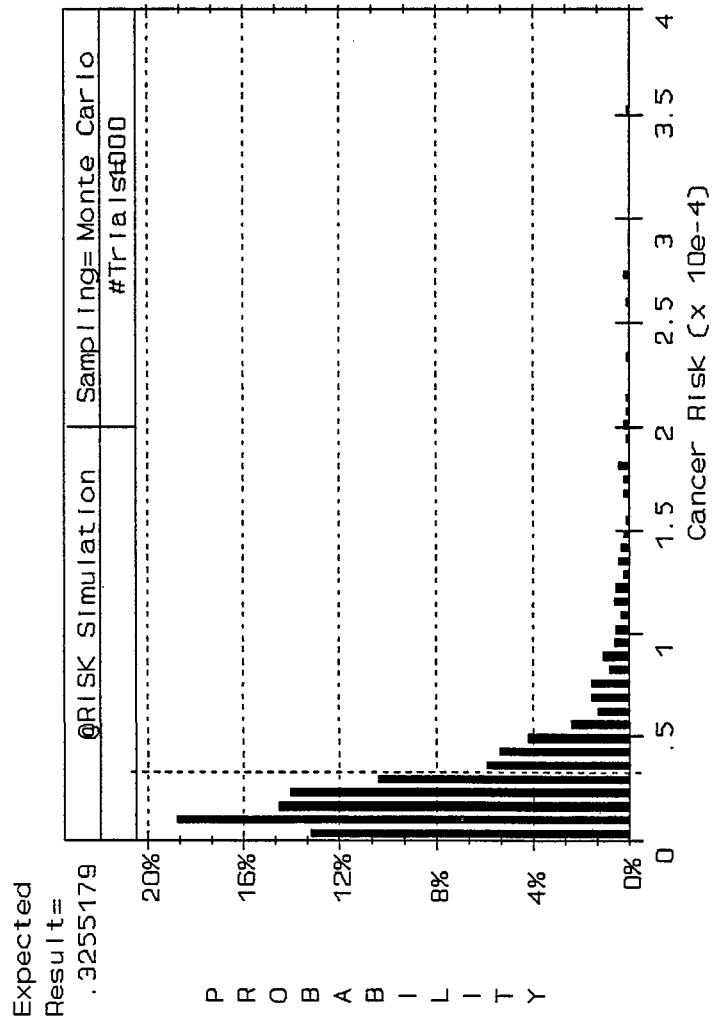


Figure 6-3  
 Excess Lifetime Cancer Risk  
 Simulation--Monte Carlo  
 North Penn Area 1  
 Phase II RI/FS

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**Section 7**  
**Summary and Conclusions**

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## Section 7 Summary and Conclusions

On the basis of the data collected, the following conclusions can be made on soil and groundwater contamination and their associated risk at the North Penn Area 1 site.

### 7.1 Soils

**Gentle Cleaners:** The chemical analyses indicated that PCE is the primary contaminant at this facility. The highest concentrations of PCE were detected in samples from the 6- to 10-foot interval in the backyard of the cleaners. One of these samples, analyzed by the CLP, had a PCE concentration of 300,000  $\mu\text{g}/\text{kg}$ . This elevated concentration suggests the presence of dense non-aqueous phase liquids sometime in the past. Concentrations were low or below detection limits in surface soils at the facility.

TCE was detected by the field GC at 1  $\mu\text{g}/\text{kg}$  in one sample at the 8- to 10-foot interval at location GC6 (Figure 4-1). No TCE was detected in the samples sent to the CLP. Acetone and methylene chloride were detected in most of the samples sent to the CLP, but the presence of these VOCs probably was the result of laboratory contamination.

The extent of soil contamination appears to be limited to parts of the backyards of the cleaners and the adjoining house.

**Granite Knitting Mills:** The highest concentration of PCE (6,900  $\mu\text{g}/\text{kg}$  by CLP) was detected at GKM-8 (Figure 4-4). Most other results were rather low with the exception of GKM-13 (1,000  $\mu\text{g}/\text{kg}$  by CLP). This location is near a storm sewer grate. Surface water samples were collected from the unnamed tributary to Skippack Creek below the outfall of this storm sewer; the samples were analyzed for VOCs by the field GC, and none were detected.

Other VOCs were detected by the field GC in GKM-8. These included trans-1,2 DCE at 175  $\mu\text{g}/\text{kg}$ , cis-1,2-DCE at 7  $\mu\text{g}/\text{kg}$ , and TCE at 53  $\mu\text{g}/\text{kg}$ ; 1,2-DCA also was detected at 45  $\mu\text{g}/\text{kg}$  by the CLP. Acetone and methylene chloride also were reported as being present in most of the CLP samples.

The results of the sampling suggest that the highest concentration of contaminants are found along the southern side of the building. The highest level of contamination in this area was found at the interval of 6 to 8 feet.

**Parkside Apartments:** The highest level of PCE observed was 1,787  $\mu\text{g}/\text{kg}$  (by field GC) at PA-6 (Figure 4-5). The highest CLP concentration was 120  $\mu\text{g}/\text{kg}$  at PA-7. The highest concentrations were found close to the building, with concentrations generally decreasing as the distance from the building increased. Cis-1,2 DCE, trans-1,2-DCE,

TCE, and 1,2-DCA also were detected at the facility. In addition, all of these contaminants were detected at sampling locations close to the building. Acetone and methylene chloride also were reported as being detected by the CLP in most of the samples.

Lexco Engineering: No VOCs were detected in any of the soil samples analyzed.

Standard Terry Mills: No VOCs were detected by the field GC in any of the soil samples analyzed. PCE, at an estimated concentration of 7  $\mu\text{g}/\text{kg}$ , was detected in one CLP sample.

## 7.2 Groundwater

The highest levels of contamination occur in wells at the center of the site, including the GKM well, NPWA wells S-9 and S-10, and the Souderton Borough well (679) (Figure 4-7). Reviewing historical data and comparing it to the results of the groundwater analyses for the RI indicate that although some VOC contamination still exists, it may have declined over time. PCE was detected at the MCL of 5  $\mu\text{g}/\text{l}$  in NPWA well S-9. PCE detections in discrete zone samples from well S-9 at concentrations from 8 to 17  $\mu\text{g}/\text{l}$  suggest that PCE levels in the aquifer around S-9 may exceed the MCL. PCE was detected at a lower concentration in well S-10 and not detected at all in well S-8. Contamination from the Area 1 plume may have affected a residential well southwest of the site. High concentrations of VOC contaminants are still entering the groundwater, at least at the location of the GKM well, although they are diluted in the aquifer to much lower levels. No solid evidence of the presence of DNAPLs in groundwater was detected, but DNAPLs would explain the presence of PCE throughout the entire depths of the GKM well and well S-9.

## 7.3 Risk Assessment

Groundwater: The overall result is that the relatively low concentrations of VOCs result in low risk levels. The excess lifetime cancer risk (current offsite residential well) for a child calculated using reasonable maximum exposure assumptions and upper-bound concentrations is about  $1 \times 10^{-6}$ . For potential future onsite residential use, the excess lifetime cancer risk for the child is  $2 \times 10^{-6}$ , while for the adult it is  $3 \times 10^{-6}$ . Potential future onsite residential use calculations were performed for groundwater infiltrating through the soils. This shallow groundwater scenario yielded an excess lifetime cancer risk for a child of  $2 \times 10^{-4}$  and for an adult of  $3 \times 10^{-4}$ .

Soil: The excess lifetime cancer risks for both the child and adult at Parkside Apartments are well below EPA's target risk level. The excess lifetime cancer risk from ingestion and dermal contact with subsurface soils at Granite Knitting Mills is at or near EPA's target level for the adult and child.

The excess lifetime cancer risk for Gentle Cleaners is at least 10 times higher than either of the other two facilities. PCE (found at 300,000  $\mu\text{g}/\text{kg}$  at the 8- to 10-foot interval) is the primary contaminant, leading to an excess lifetime cancer risk of  $4 \times 10^{-5}$  for the child and  $3 \times 10^{-5}$  for the adult. However, the average risks calculated are about an order of magnitude lower than the calculated reasonable maximum risk levels.

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**Section 8**  
**References**

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## Section 8 References

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